Environmental Management
DOI 10.1007/s00267-012-9970-y

Ecological Engineering Practices for the Reduction of Excess Nitrogen in Human-Influenced Landscapes: A Guide for Watershed Managers

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Received: 9 January 2012 / Accepted: 1 September 2012

Abstract Excess nitrogen (N) in freshwater systems, estuaries, and coastal areas has well-documented deleterious effects on ecosystems. Ecological engineering practices (EEPs) may be effective at decreasing nonpoint source N leaching to surface and groundwater. However, few studies have synthesized current knowledge about the functioning principles, performance, and cost of common EEPs used to mitigate N pollution at the watershed scale. Our review describes seven EEPs known to decrease N to help watershed managers select the most effective techniques from among the following approaches: advanced-treatment septic systems, low-impact development (LID) structures, permeable reactive barriers, treatment wetlands, riparian buffers, artificial lakes and reservoirs, and stream restoration. Our results show a broad range of N-removal effectiveness but suggest that all techniques could be optimized for N removal by promoting and sustaining conditions conducive to biological transformations (e.g., denitrification). Generally, N-removal efficiency is particularly affected by hydraulic residence time, organic carbon availability, and establishment of anaerobic conditions. There remains a critical need for systematic empirical studies documenting N-removal efficiency among EEPs and potential environmental and economic tradeoffs associated with the widespread use of these techniques. Under current trajectories of N inputs, land use, and climate change, ecological engineering alone may be insufficient to manage N in many watersheds, suggesting that N-pollution source prevention remains a critical need. Improved understanding of N-removal effectiveness and modeling efforts will be critical in building decision support tools to help guide the selection and application of best EEPs for N management.

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Published online: 22 November 2012

Keywords Ecological engineering · Nitrogen removal · Watershed · Land use · Aquatic ecosystems · Decision making

Introduction

Excess nitrogen (N) in freshwater systems, estuaries, and coastal areas is responsible for water-quality degradation, eutrophication, and hypoxia in some of the most ecologically and economically important water bodies in North America and elsewhere, including the North Sea, Gulf of Mexico, and Chesapeake Bay (Diaz and Rosenberg 2008; Martin and others 1999; Rabalais and others 1996, 2001). Consequently, many studies over the last 40 years have focused on approaches to mitigate the impact of N from human activities (e.g., agriculture, urbanization) on water quality (Carpenter and others 1998; Craig and others 2008; Dietz 2007; Dosskey 2001; Gold and Sims 2000; Howarth and others 2000; Kadlec 2009; Robertson and others 2000). Two complementary approaches have emerged: (1) N-source control and reduction and (2) interception and treatment of sources of N.

Source-reduction strategies [approach no. 1 (see above)] include all practices whose primary objective is to decrease N inputs into landscapes, including limits on fertilizer or manure application to lawns and agricultural lands, controls on atmospheric N deposition from fuel combustion, and decreasing N wastewater discharge from urban areas. However, regardless of source control or reduction, a portion of N input will leak from catchments by way of sewage infrastructure, agricultural ditches, tile drains, runoff from impervious surfaces (roads, parking lots), and septic systems. Complementary ecological engineering approaches are therefore needed to intercept and decrease N leaching to aquatic environments [approach no. 2 (see above)].

This review addresses ecological engineering practices (EEPs) aimed at intercepting and decreasing the transfer of N from upland environments (primarily urban/suburban development and agricultural areas) to aquatic environments, including groundwater, surface water, and ultimately coastal waters. In this context, we broadly defined ecological engineering as the design of ecosystems for the mutual benefit of humans and nature (Mitsch 1992). Some common EEPs include advanced-treatment septic systems, low-impact development (LID) designs, permeable reactive barriers (PRBs), treatment wetlands, riparian buffers (when actively managed or engineered), artificial lakes and reservoirs, and stream restoration (Fig. 1). The primary purpose of these EEPs is not always N mitigation; however, the design of these systems can often be optimized for N removal (Collins and others 2010a; Craig and others 2008; Dietz 2007; Dosskey 2001; Gold and Sims 2000; Kadlec 2009; Robertson and others 2000). We therefore propose that the potential of these EEPs for N removal should be taken into account when developing whole watershed–management strategies.

Over the years, a significant amount of knowledge has been generated on the design and functioning principles of some of these EEPs. Although N assimilation by plants and microorganisms, dissimilatory nitrate reduction to ammonia, and anaerobic ammonia oxidation may contribute to N retention in some systems (Burgin and Hamilton 2008), respiratory denitrification, or the microbial transformation of nitrate nitrogen (NO_3^-) to N_2 and N_2O gases, is considered to be the most substantial and important N-removal process in many EEPs (Saunders and Kalff 2001). Several studies have attempted to synthesize knowledge about processes regulating the N-removal efficiency of some of these EEPs, including studies that report N-removal efficiencies for managed riparian buffers (Mayer and others 2007; Zhang and others 2010), bioreactors (Schipper and others 2010a), or LID systems (Collins and others 2010b). Others have described how "hot spots" and "hot moments" of biogeochemical transformation and/or transport can contribute to the removal of N or other contaminants in the landscape (Groffman and others 2009; Vidon and others 2010). Several studies have also addressed the critical question of EEP placement in landscapes to optimize N-removal benefits at the watershed scale (Dosskey and Qiu 2010; Kellogg and others 2010). However, no studies to date have attempted to synthesize current knowledge about the functioning principles, performance, and cost of multiple commonly used EEPs to help managers develop more efficient N-mitigation programs at the watershed scale. More importantly, no studies identify where and when each of these EEPs should be implemented for maximum environmental benefits. We believe that this lack of summary information for a range of EEPs limits the ability of watershed managers to make informed decisions about the relative cost and N-removal performance and effectiveness of various approaches based on local conditions when developing watershed-management plans.

In this review, we summarize the current understanding of the functioning principles, performance, and cost of implementation available for seven important EEPs with significant potential to decrease N across terrestrial and/or aquatic environments. We then discuss how to prioritize EEP selection and placement in a watershed depending on the primary N-pollution source. Often, critical knowledge is missing, especially because it relates to cost of implementation or N-removal efficiencies among physiographic regions. When possible, we identified gaps in knowledge. This review is a first step toward the development of decision support tools for advanced scenario modeling incorporating multiple N-management practices at the

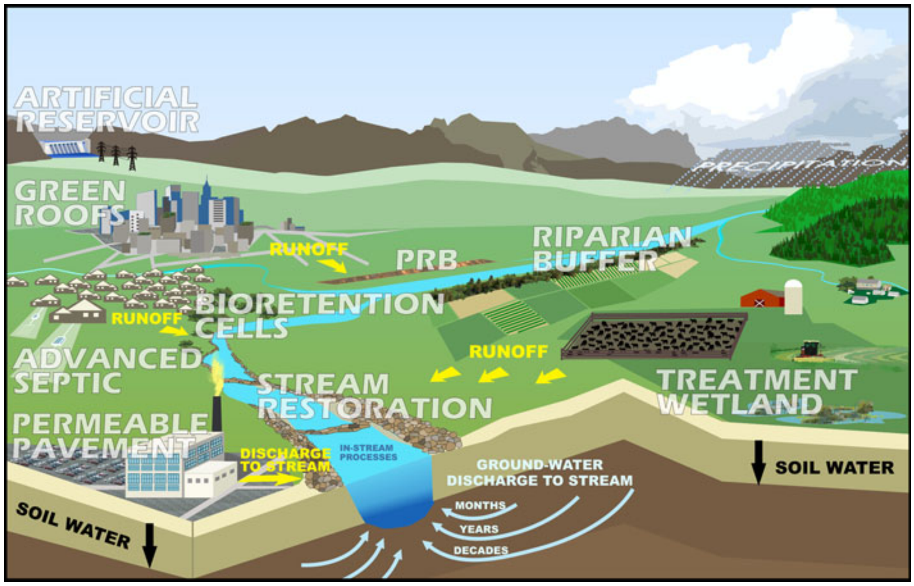

Fig. 1 Sources and pathways of N, including urban, industrialized and agricultural areas, in a conceptualized watershed. N may follow atmospheric, surface, and subsurface pathways at various spatial and temporal scales. EEPs may be applicable to specific or multiple sources and some may be used in combination to address one or more N pathways. See text for further explanation

watershed scale. Here we review the following: advanced-treatment septic systems (approach no. 1), LID structures (approach no. 2), PRBs (approach no. 3), treatment wetlands(approach no. 4), managed riparian buffers (approach no. 5), artificial lakes and reservoirs (approach no. 6), and stream restoration (approach no. 7). We organized these EEPs based on their placement from upland (approaches no. 1–3), to terrestrial-aquatic transition areas (approaches no. 4 and 5), to the aquatic environment (approaches no. 6 and 7).

Approach No. 1: Advanced-Treatment Septic Systems

Conventional septic systems using a septic tank/soil absorption system have been in use for decades to collect domestic wastewater and release it to the subsurface environment. These systems generally discharge septic tank water in the soil below the root zone through a drain field or lateral drains (Gold and Sims 2000). These systems are not designed to decrease N loading and are often point sources of N in the landscape, abating only 10–20 % of N loads (Keeney 1986; Lamb and others 1990; Siegrist and Jenssen 1989). Therefore, advanced-treatment septic systems have been developed to lower N load in wastewater before release to the environment (Fig. 2). These advanced systems enhance biological N-removal processes through a series of steps designed to promote nitrification and denitrification (Oakley and others 2010), first through aeration to support oxidation of ammonium to nitrate, then by adding labile carbon (C) to decrease nitrate into N gases by way of denitrification, thereby effectively removing N from the wastewater (Gold and Sims 2000).

Two subtypes of advanced-treatment septic systems exist. Recirculating systems recycle wastewater through the nitrification/denitrification steps using organic matter in the wastewater as a source of C to fuel denitrification (Oakley and others 2010). Sequential systems employ nitrification and denitrification sequentially, and use supplemental C, such as wood chips, to fuel heterotrophic microbial processes (Oakley and others 2010; Schipper and others 2010a). Because advanced septic systems also help achieve lower biological oxygen demands and lower total suspended solid concentrations, it is possible to disperse the effluent higher in the soil profile (closer to the root zone) without increasing the risk of hydraulic failure (i.e., clogging and subsequent surface ponding) and allow for further N removal as the effluent percolates through the soil profile.

The N-removal effectiveness of advanced septic systems varies widely depending on the specific design, maintenance, environmental conditions, number of people supported, and whether the system is used continuously or seasonally. Oakley and others (2010) summarized data from three separate field studies (Florida, OR, New Zealand) examining the performance of 20 advanced-treatment septic systems. N-removal efficiencies for recirculating systems ranged from 40 to 70 % of the N load (organic and inorganic N combined). Sequential systems using wood chips exhibited N-removal efficiencies >90 % (Table 1). In recirculating systems, a part of the effluent does not go through the denitrification unit. In addition, some free oxygen from the aerated unit might be transferred to the anaerobic reactor where denitrification efficiency may be decreased.

The cost of installing advanced-treatment septic systems varies as a function of size, soil type, and design but

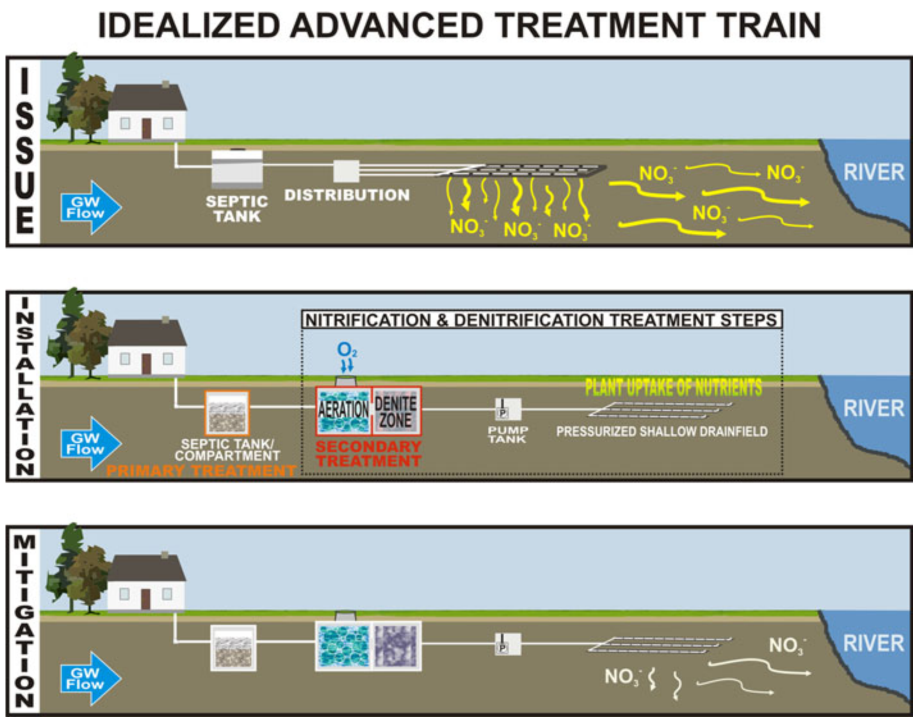

Fig. 2 Advanced-treatment septic system. (*Issue*) Advanced treatment systems are designed for treating N from residential wastewater. (*Installation*) Such systems incorporate a secondary treatment step between solids separation (primary treatment) and final dispersal of effluent. Pumps, timers, and floats are used to control the flow of wastewater from one component to the next. Secondary treatment includes an aerobic and an anaerobic denitrification ("denite") zone functioning either in sequential or in recirculating modes. (*Mitigation*) Effluents containing lower N concentrations reach groundwater (GW) and receiving waters

generally ranges from $15,000 to $30,000, including design and installation. As a comparison, conventional gravity system installation costs vary from $13,000 to as much as $45,000 depending on soil characteristics. Conventional gravity systems also require larger drainfields than advanced-treatment systems. Advanced-treatment systems modulate flow to the drainfield with peak flows stored and released at a consistent rate, thus allowing for more effective N removal. Conventional systems accommodate peak flows in the drainfield, requiring more space and resulting in inconsistent N removal. Spreadsheet-based models have been developed to estimate cost based on site characteristics and design (Water Environment Research Foundation 2010). After initial installation, regular operation and maintenance is necessary to continuously achieve high levels of wastewater treatment and N removal. Costs of operation and maintenance are influenced by electrical energy demands of the system. Media filters, which provide surface area for bacteria to colonize and allow for biochemical and physical treatment processes to occur, add approximately $100 annually to the cost. Systems with continuously operating fans that promote aeration during secondary treatment may add three to four times this cost to the electric bill for a typical three bedroom home. Most technologies typically require two maintenance visits per year ($200–$600/year). Site characteristics (e.g., soils, slope, available land area) and system requirements (e.g., number of people served) vary widely and dictate design, thus making it impossible in this review to assess the relative merits of the many advanced-treatment system configurations. A system that is most effective in terms of cost and N removal for one site will be different from that for another site.

Approach No. 2: LID Structures

LID practices encompass approaches to land development that attempt to mimic natural systems to manage stormwater primarily in urban or suburban environments (United States Environmental Protection Agency [USEPA] 2011; Mitsch 1992). Specific LID techniques include, but are not limited to, green roofs, bioretention cells, and permeable pavement systems (Figs. 3, 4, 5). Most LID practices are generally designed to decrease the volume of stormwater runoff to drainage systems and streams by way of interception, evapotranspiration, and infiltration, thus disconnecting impervious surfaces from the conventional stormwater network. The primary intent is not to remove N but to decrease peak flows and overall runoff water volumes to receiving waters to alleviate streambank erosion and altered hydrologic patterns associated with urban/suburban land cover (Dietz 2007; Meyer and others 2005; Thurston and others 2008; Walsh and others 2005). Green roofs, which consist of a shallow layer of lightweight media, such as expanded shales and clays that support a dense cover of drought-resistant, herbaceous vegetation (Fig. 3), achieve this reduction through the direct

Table 1 Type, subtype (i, ii, iii), NO_3^- removal efficiency (as a percentage of N input), and cost for seven commonly used EEPs

Type	Subtype	NO_3^- removal (%)[a]	Cost	Remarks	References
Advanced septic systems	Recirculating systems (i); sequential systems (ii)	40–70 % (i); >90 % (ii)	\$15–\$30 K (includes design and installation)	Additional costs: \$100/year for media filter and \$200–\$600/year for maintenance	Oakley and others (2010) and Water Environment Research Foundation (2010)
LID techniques	Green roofs (i); bioretention cells (ii); permeable pavement (iii)	68 % (0–96 % (i); −61 % (−650 to 85 %) (ii); –97 % (−331 to 69 %) (iii)	\$86–\$161/m^2 (i); \$43–\$430/m^2 (ii); \$22–\$108/m^2 (iii)	Systems generally not designed for N removal, but designs can be optimized to achieve greater N removal	Collins and others (2010a), Dietz (2007), LID Center (2011a, b)
PRBs	Denitrification walls (i); denitrification beds (ii); denitrification layers (iii)	>90 % (i); >90 % (ii); >90 % (iii)	Variable: \$2–\$15/kg N removed	50 % initial nitrate removal rate after 15 years	Robertson and others (2008) and Schipper and others (2010a)
Treatment wetlands	FWS wetlands (i); horizontal sub-SWF wetlands (ii); vertical sub-SWF wetlands (iii)	40–44 % (mean); (0–100 % depending on design)	Variable: \$0.001–\$0.1/m^2 (i); \$0.03–\$1/m^2 (ii)	Works best with C:N >5:1 and permanently saturated conditions plus long residence time (e.g. type ii)	Hammer and Knight (1994), Kadlec and Wallace (2008), Mitsch and others (2005) and BMP database (2010)
Riparian zones	Coarse sand and gravel soil (i); other soil (ii)	40–100 % (i); 90–100 % (ii)	Variable: e.g., \$0.0262/m^2	70–90 % removal may be achieved in ≥25 m in many riparian zones	Mayer and others (2007), Roberts and others (2009) and Vidon and Hill (2006)
Artificial lakes and reservoirs	None	10–100 %	A few thousands of dollars to millions of dollars	N removal rate strongly positively correlated to residence time	Forshay and Stanley (2005), Kellogg and others (2010) and Wall and others (2005)
Stream restoration	Organic matter additions (i); channel reconstruction (ii); floodplain reconnection (iii); artificial geomorphic features (iv); bank stabilization (v)	5–40 %; 11 % during base-flow; 24 % during high flows	\$15–\$812 K/project; \$520–\$1526/m restored stream	Many restored streams contain "hot spots" of N removal, but overall effect on water quality with respect to N at the reach scale remains uncertain	Bernhardt and others (2005), Filoso and Palmer (2011), Kaushal and others (2008a), Mayer and others (2010) and USEPA (2006)

LID low impact development, *PRBs* permeable reactive barriers

[a] N-removal efficiencies are only expressed as a percent removal because mass removal metrics available in the literature do not allow for a direct comparison between EEPs in terms of mass N removed

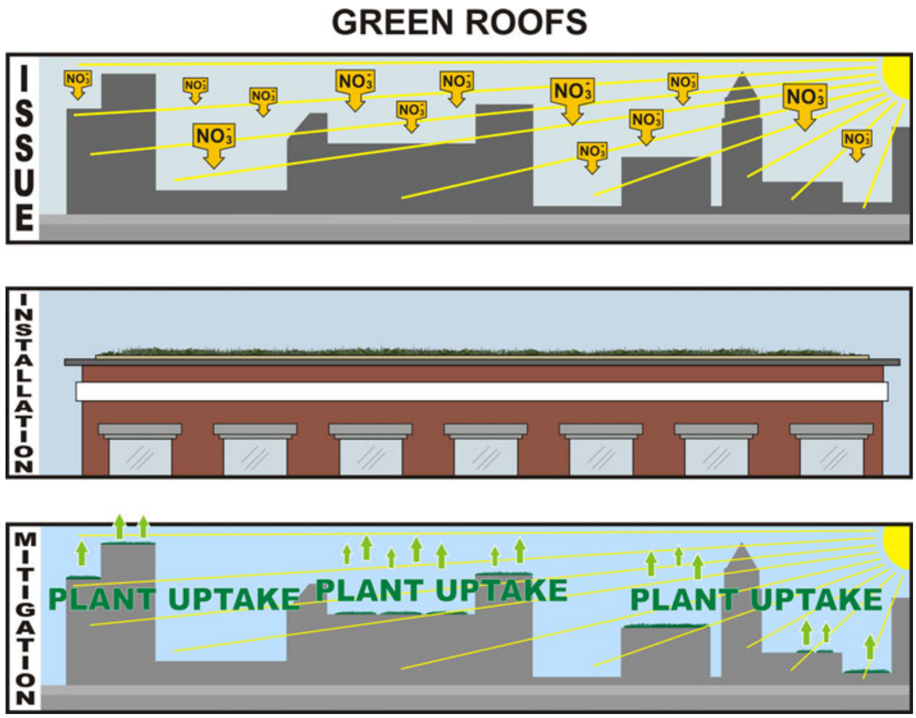

Fig. 3 Green roofs. (*Issue*) N in the atmosphere falls on bare roofs and flows untreated to stormwater systems and/or receiving water bodies. (*Installation*) Green roof system with short grass vegetation. (*Mitigation*) N is intercepted by green roofs and mostly undergoes plant uptake. In less frequent cases (e.g., deeper soil media, taller vegetation), some N may be returned to the atmosphere in gas form if conditions in the contained soils are conducive to microbial denitrification

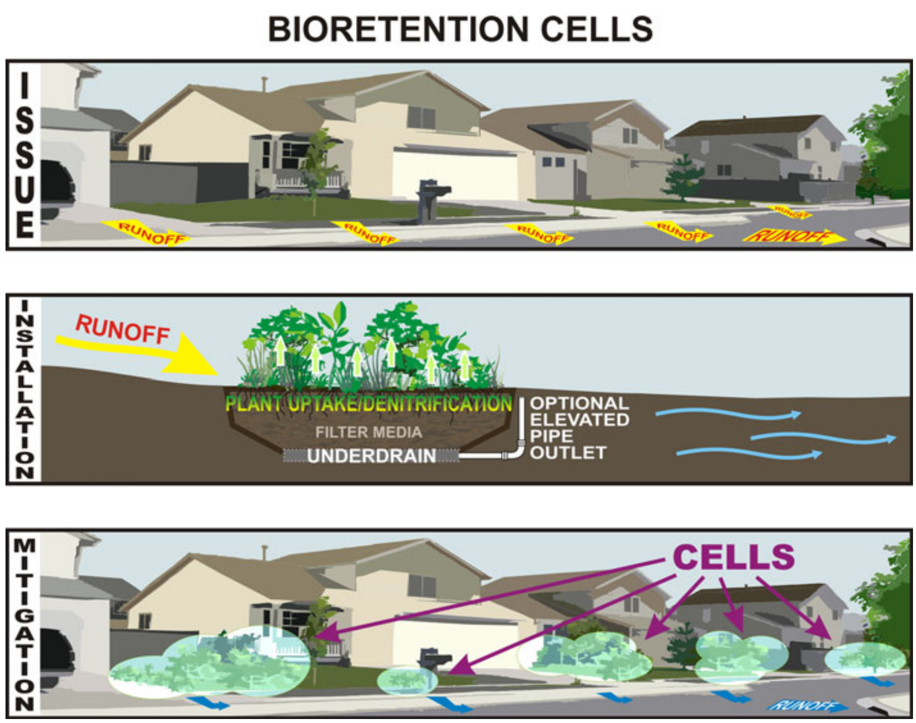

Fig. 4 Bioretention cells. (*Issue*) On impervious surfaces, N would normally run off toward sewerage after a rainfall event. (*Installation*) Bioretention system with optional elevated pipe outlet. (*Mitigation*) N may be captured by bioretention systems and assimilated by plants and soil microbes

interception and evapotranspiration of precipitation (Dietz 2007). Bioretention systems, also referred to as “rain gardens” or “bioswales,” are shallow depressions containing soil filter media that support drought- and flood-resistant vegetation and achieve stormwater runoff reductions through the interception and infiltration of runoff from impervious surfaces through the media to an underdrain or underlying soil (Dietz 2007) (Fig. 4). Permeable pavement systems consist of a variety of paving surfaces containing void spaces that allow for stormwater infiltration through aggregate sublayers to an underdrain or underlying soil (Dietz 2007) (Fig. 5).

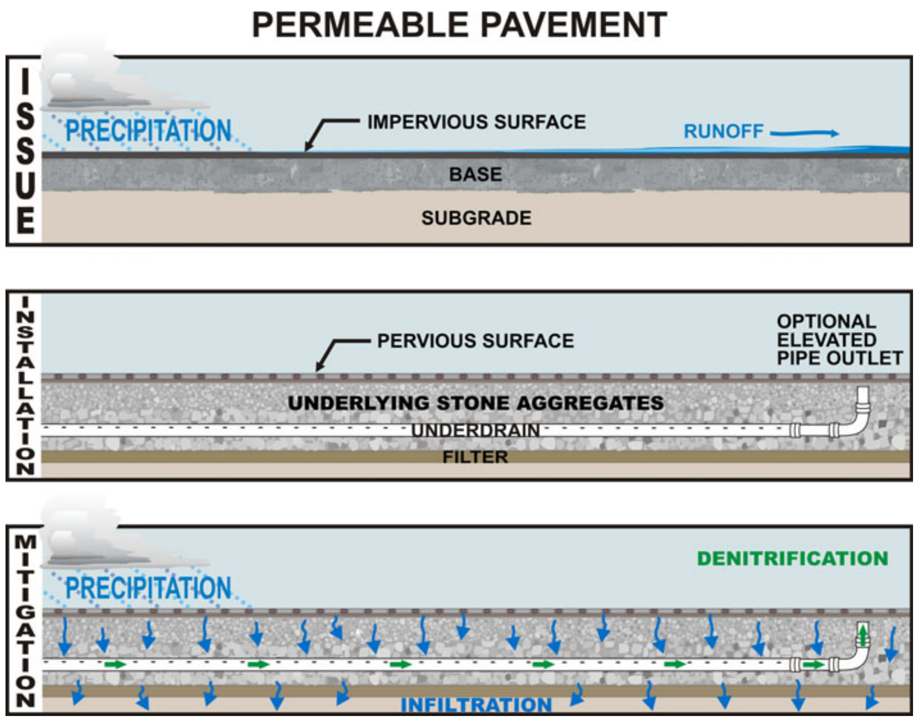

Fig. 5 Permeable pavements. (*Issue*) On impervious surfaces, N would normally run off toward sewerage after a rainfall event. (*Installation*) Permeable pavement overlying a porous media (e.g., gravel, stone aggregates) and an underdrain is shown. The underdrain intercepts percolating water. In this diagram, an optional elevated pipe at the underdrain outlet is proposed to enhance the development of decreasing conditions in the soil media. (Mitigation) Runoff infiltrates through the permeable pavement soil media, and N concentrations are decreased

Recently, attention has been focused on the potential pollutant-removal mechanisms in LID structures, including adsorption, sequestration, or transformation due to chemical, physical, or biological processes during interception and infiltration (Dietz 2007). To increase the volume of influent runoff to be intercepted, rapid drainage is needed. However, this in turn decreases opportunities for the development of anaerobic zones conducive to denitrification. Aerobic conditions can also support organic matter mineralization and nitrification and thus generate an increase in both ammonia and nitrate in runoff (Hsieh and others 2007).

Consequently, although LID practices can be optimized to decrease N, such reduction cannot generally be achieved without increasing the water residence time in LID structures, thus leading to a wide range of N-removal efficiency for LID systems. For instance, a recent review of N-removal performance across LID types (Collins and others 2010a) found that N removal ranges from 0 to 96 % (median 83 %) for nitrate and −311 to 91 % (median 7 %) for total nitrogen (TN) in green roofs in North Carolina, Sweden, and Japan. In bioretention cells in North Carolina, Maryland, Australia, and bench-scale systems in laboratories and greenhouses in multiple other locations, nitrate and TN removal range from −650 to 85 % (median 8 %), and −312 to 58 % (median 25 %), respectively. Finally, nitrate removal of −331 to 69 % (median −59 %), and TN removal ranging from 42 to 91 % (median 50 %), were observed in permeable pavements in North Carolina, Connecticut, and France.

Dominant N-removal processes also vary depending on the LID structure type. In green roofs, plant uptake is the major N-removal mechanism (Czemiel Berndtsson and others 2006). Indeed, the thin soils of most green roofs typically are not designed to provide extended periods of anaerobic conditions conducive to denitrification. Increasing soil media depth can improve N removal by providing opportunities for planting taller vegetation, which often exhibits a larger N-retention capacity than grass or moss (Czemiel Berndtsson and others 2009). However, taller vegetation often requires more frequent maintenance to sustain plant growth, including the use of N-based fertilizers (Czemiel Berndtsson and others 2006). In bioretention cells, vegetated systems typically remove more N than nonvegetated systems (Lucas and Greenway 2008; Read and others 2008). However, a recent study by Passeport and others (2009) did not indicate that vegetation type had a significant effect on N removal in these systems. Anaerobic conditions conducive to denitrification can be engineered into bioretention cells and permeable pavements through the elevation of outlet pipes in systems with underdrains (Collins and others 2008; Dietz and Clausen 2006; Kim and others 2003; Passeport and others 2009), or through the use of fine-textured media layers (Cho and others 2009; Hsieh and others 2007; Hunt and others 2006). Shallow sand layers may also be incorporated to provide additional surface area for microbial colonization and N removal (Collins and others 2010b).

Common key design parameters for green roofs, bioretention cells, and permeable pavements for enhancing N removal therefore include the following: (1) the establishment of low nutrient–demanding vegetation to limit the need for fertilization, favor plant N uptake, and provide a continuous supply of organic matter by way of plant decomposition; (2) designs that increase water residence time to enhance interactions between N and microbial populations; and (3) the introduction of design elements conducive to the development of anaerobic conditions (e.g., elevated underdrains, fine-textured soil media). However, tradeoffs must be found to limit export of vegetation, which can be a source of N in the effluent, and to maintain LID systems' first objective (runoff water volume reduction) while simultaneously maintaining anaerobic conditions.

Bioretention cells are relatively easy and inexpensive to construct and maintain with typical costs running up to \$43/m^2 (\$4/square foot) in a residential application and between \$108 and \$430/m^2 (\$10 and \$40/square foot) in an institutional application (LID Center 2011a). Costs may increase where clay soils impede infiltration and where additional soil media must be purchased and installed to increase infiltration capacity. Maintenance requirements include occasional watering and replanting to maintain vegetation. It also requires the periodic replacement of the top several centimeters of media to prevent clogging by fine suspended solids and replenish the soil organic matter pool. Green roofs and permeable pavement systems are typically more expensive to construct and maintain costing approximately \$86–\$161/m^2 (\$8–\$15/square foot) for all green roof applications and \$22–\$108/m^2 (\$2–\$10/square foot) for permeable pavement systems (LID Center 2011b). Green roofs require occasional watering and replanting, whereas permeable pavement systems require periodic sweeping or vacuuming to remove accumulated solids that may cause clogging.

Overall, green roofs, bioretention cells, and permeable pavement systems present similar cost/benefit ratios in terms of N removal. However, bioretention cells and permeable pavement systems engineered to maintain saturated anaerobic conditions have greater N-removal capacity than green roof systems. Bioretention and permeable pavement systems can store more water on a square-meter basis than green roofs and therefore have the ability to treat larger runoff volumes than green roofs. However, these systems are not implemented in the same types of locations in watersheds and could be used in concert when cost allows. More research is needed to properly quantify N-removal performance among LID systems and identify strategies to modify current LID designs to enhance N removal without negatively affecting their ability to intercept stormwater and remove other pollutants of concern, such as phosphorus (P) and heavy metals. It is also important to differentiate N plant uptake (temporary removal unless vegetation is harvested) from N removal by way of denitrification (permanent removal) in LID systems. This requires moving beyond N mass balance approaches and conducting direct measurements of *in situ* N cycling in these systems (Collins and others 2010a).

Approach No. 3: PRBs

PRBs are constructed zones of reactive material that extend below the water table to intercept and treat contaminated groundwater (Fig. 6). Such barriers have been used to treat various contaminants, such as chlorinated solvents, chromium, arsenic, and organic and inorganic compounds (He and others 2008; Ludwig and others 2009; Wilkin and others 2009). More recently, PRBs have been employed to remediate N pollution from surface water and shallow groundwater (Robertson and Cherry 1995; Robertson and others 2000; Schipper and Vojvodic-Vukovic 1998, 2000, 2001). PRBs for N-removal function by creating a subsurface environment favorable to denitrification.

PRBs, also called "biowalls" or "bioreactors," are typically installed by digging a trench designed to intercept the flow of contaminated groundwater and are most effective when the source of contamination is concentrated in a plume (Fig. 6). The trench is then filled with organic matter to serve as a C source for heterotrophic bacteria, such as sawdust, wood chips, or straw and/or reactive materials, such as iron or sulfur. Sand may be mixed with the reactive material to increase permeability. An impermeable wall may be added to direct the groundwater flow toward the reactive parts of the barrier. The PRB wall is usually then covered with soil. Numerous subtypes of PRBs exist that vary by C source, installation, and incorporation into the substrate (Schipper and others 2010a). "Denitrification walls" are installed vertically into the subsurface perpendicular to groundwater flow. "Denitrification beds" are containers (usually 1–2 m deep in varied lengths and widths) filled with organic matter that receive discharge from wastewater or agricultural drainage, whereas "denitrification layers" are horizontal layers of organic material incorporated into unlined or unconsolidated subsurface sediments. This latter approach may involve deep-soil mixing with augers or other processes that create vertical treatment zones to treat deeper plumes. PRBs work best for treating nitrate in shallow (4–5 m deep) concentrated zones where contaminated groundwater moves in plumes in a focused direction that allows the targeted placement of the wall, bed, or layer of organic matter.

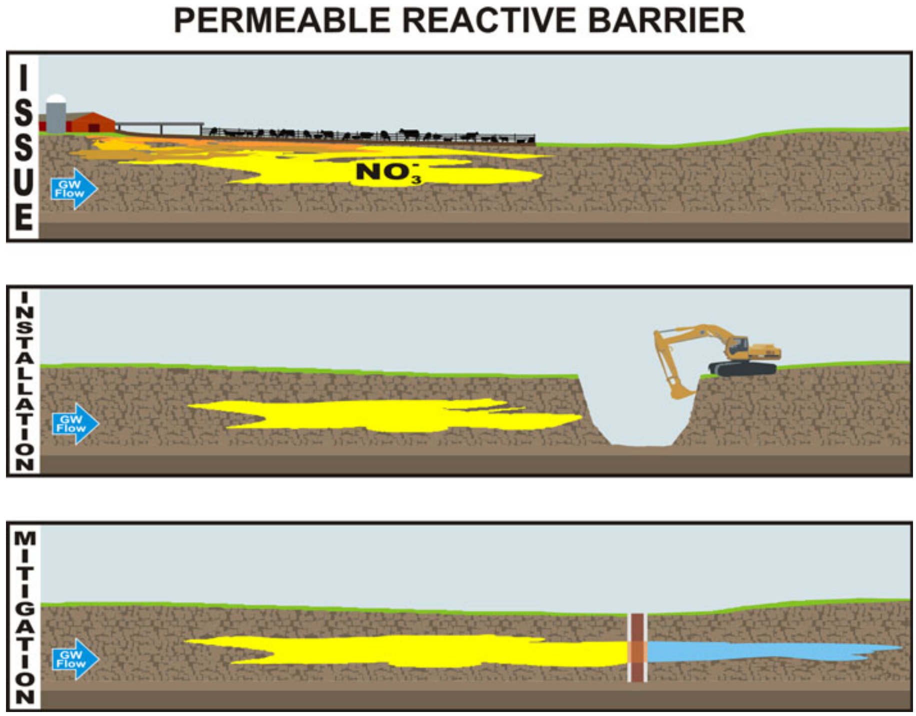

Fig. 6 PRB depicted as a management approach for concentrated N in groundwater (GW). (*Issue*) N from a known source such as an animal feeding operation may accumulate in groundwater. (*Installation*) The PRB is constructed to intercept subsurface flow. (*Mitigation*) N in the groundwater contacts organic substrates in the barrier where denitrification by microbes may remove N

To be effective, PRBs must be positioned in a manner where subsurface flow intersects with the reactive portion of the wall. To optimize contact with subsurface flow, Schipper and others (2010a) recommended the installation of PRBs no deeper than 4–5 m in permeable media that allows for adequate flow rates. The presence of a confining layer below this media also encourages contact between PRB material and groundwater flow paths. Although guidelines for denitrifying PRBs have not yet been standardized, designs should address basic hydrology, size, and flow rates of the contaminated plume, N concentrations, and seasonal flow variability (Schipper and others 2010a). For example, the seasonality of groundwater conditions should be taken into account so that the depth of the barrier is suitable for high and low water table conditions. Similarly, soil permeability must be taken into account. Although high-permeability PRBs may decrease contact time between N-laden groundwater and the substrate, PRBs with high permeability may also serve to create flow convergence and upwelling in the direction of the wall, thereby increasing contact with the wall substrate (Robertson and others 2005).

N-removal processes in PRBs may include immobilization, dissimilatory nitrate reduction to ammonium (DNRA), and/or anaerobic ammonium oxidation (anammox); however, heterotrophic denitrification is believed to be the dominant N-removal mechanism in these systems (Schipper and others 2010a). Therefore, N removal in PRBs depends on the conditions that foster denitrification including anoxic subsurface conditions, availability of C, nitrate concentration, temperature, and groundwater flow paths and flow rates. Sources of C in PRBs designed for N removal usually are inexpensive and widely available (e.g., wood chips, sawdust, etc.), may remain effective for years, and require minimal maintenance or replacement (Robertson 2010; Robertson and others 2000, 2008, 2009; Schipper and Vojvodic-Vukovic 2001; Schipper and others 2005). Vegetable oil, cotton seed burrs, and molasses have been used to create denitrifying barriers, but these materials are more expensive than wood chips or sawdust and require more frequent substrate replacement or replenishment (Hunter 2001; Su and Puls 2007; Schipper and others 2010a). Although there have been no documented cases of PRB failures due to C limitation, should failure occur, replacement of the PRB would likely involve either excavation of the existing barrier and fill replacement or the installation of a new barrier. Because nitrate acts as an electron acceptor during the heterotrophic denitrification process, the efficiency of PRBs may be limited by low rates of nitrification. Conversely, denitrification beds may be overwhelmed if nitrate concentrations are extremely high, although removal rates may still be significant (Schipper and others 2010b).

With respect to efficiency, N removal from groundwater using PRBs is generally high and often exceeds 90 % (Robertson and Cherry 1995; Robertson and others 2000; Schipper and Vojvodic-Vukovic 1998, 2000, 2001). In some cases, PRBs have shown complete N removal from wastewater effluents containing $\leq$250 g NO_3^--N m^{-3} (Schipper and others 2010a). However, high variability of N mass removed is observed among studies with removal ranging from 0.62 to 12.7 g NO_3^--N m^{-3} day^{-1} (median removal rate 2.5 g NO_3^--N m^{-3} day^{-1}) (Schipper and others 2010a). Several long-term studies have observed efficient functioning of PRBs for $\leq$15 years (Moorman and others 2010; Robertson and Cherry 1995; Robertson and others 2008). For instance, in a PRB in Ontario, Canada, Robertson and others (2008) observed N-removal rates of 4.6 $\pm$ 0.7 g NO_3^--N m^{-3} day^{-1} after 15 years. These rates were approximately 50 % of the initial N-removal rates (10.2 $\pm$ 2.7 g NO_3^--N m^{-3} day^{-1}). Granger and others (2007) used results from laboratory experiments measuring N removal using woodchips, combined with stoichiometric assumptions for C:N consumption, to predict the life expectancy of woodchip PRBs, which ranged from 30 to >100 years depending on nitrate concentrations. In a separate study, Moorman and others (2010) found the half-life of organic matter substrates to vary within a barrier from 4 to >36 years depending on C consumption, with portions of the barrier remaining saturated most of the time, but estimated to last much longer due to decreased decomposition of the C-fill material under anaerobic conditions (Moorman and others 2010).

Principal costs associated with PRBs are for excavation and for organic material to fill the wall or bed. Construction costs vary depending on the size, design, and C source. Half-life of the substrate will determine longevity and subsequently the long-term cost. Cost estimates per unit mass of N removed range from approximately \$2 to \$15 (USD)/kg N over the lifetime of the barrier. Cost is comparable with other N-management systems, such as constructed wetlands (Schipper and others 2010a) (Table 1).

Approach No. 4: Treatment Wetlands

Both natural (Fisher and Acreman 2004; Jordan and others 2011; Lowrance and others 1995) and constructed (Hernandez and Mitsch 2007; Kadlec 2009; Mitsch and others 2005; Vymazal and others 2006) wetlands have been widely studied for their ability to remove N from agricultural (Braskerud 2002; Tanner and others 2005), municipal, and industrial wastewaters (Hammer 1989; Vymazal 2005, 2009) (Fig. 7). Constructed wetlands have often been classified according to water flow regime: free water surface (FWS), horizontal subsurface flow (HSSF) or vertical subsurface flow (VSSF) wetlands (Kadlec and Wallace 2008) (Fig. 8). Contrary to HSSF and VSSF wetlands, where the water level is maintained below the soil surface, FWS wetlands present open water areas and are often classified based on vegetation type (i.e., floating, submerged, and emergent). In HSSF wetlands, water flows horizontally from a point inlet structure to an outlet one. In VSSF wetlands, the inlet structure is designed to distribute

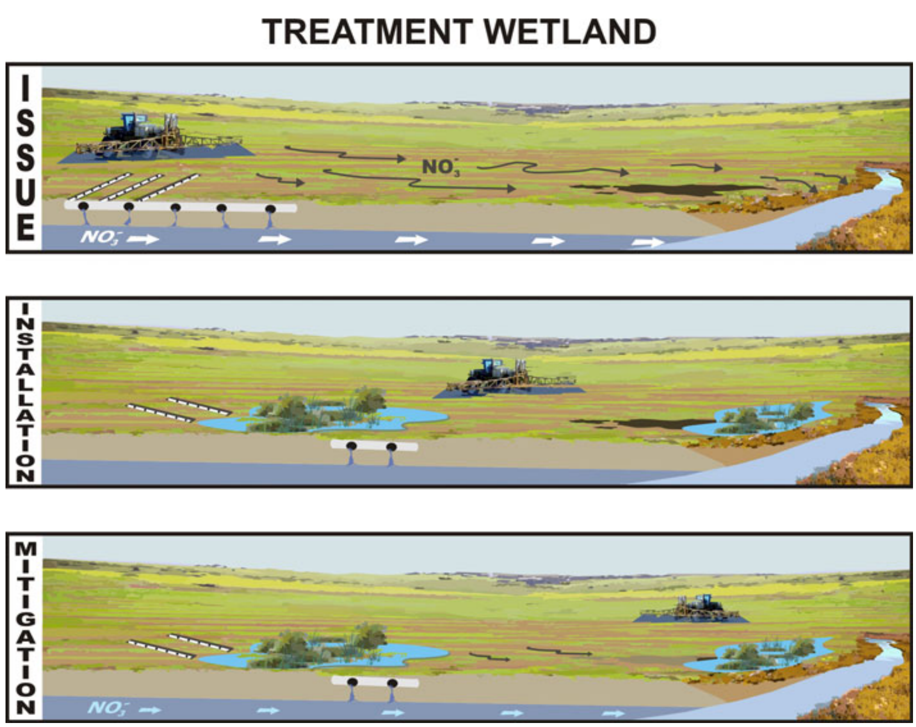

Fig. 7 Treatment wetlands. (*Issue*) In agricultural areas, fertilizer application results in high N concentrations in surface runoff and subsurface tile drain flows connected to receiving rivers. (*Installation*) Placement of treatment wetland in contaminated flows for N interception. (*Mitigation*) Aquatic vegetation and algae may assimilate N, and microbes in the wetland sediments may remove N through denitrification

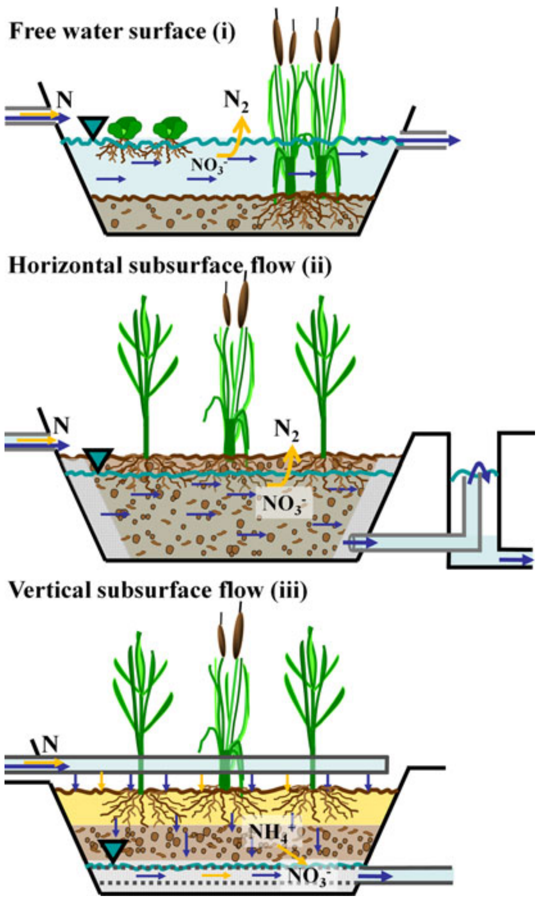

Fig. 8 Main types of artificial treatment wetlands based on hydraulic functioning class; (i) FWS wetlands have open water areas with floating, submerged or emergent vegetation; (ii) HSSF wetlands: water flows horizontally in the subsurface from inlet to outlet structures passing through a media made of soil and gravel; optional upturned outlet pipe can help the development of anaerobic conditions favorable to denitrification; (iii) VSSF wetlands also include a soil/gravel media through which water flows as subsurface flow. Contrary to HSSF wetlands, the water inlet structure is designed to distribute influent over the entire wetland surface

the water evenly over the entire wetland surface. Water then percolates through the soil media and is collected in a subsurface underdrain. Both HSSF and VSSF wetlands soil media generally consists of coarse material (e.g., gravel), which provides physical support for plants, surface area for chemical reactions, and microbial population development (Hammer 1992; Albuquerque and others 2009). The range of designs reflects the multiple hydrological functions of these systems and their associated effects on N cycling. Heterotrophic denitrification is often the dominant N-removal process in treatment wetlands, although plant uptake combined with vegetation harvesting to permanently remove N from the system can significantly contribute to N removal (Vymazal and others 2006).

Published denitrification rates in treatment wetlands vary over seven orders of magnitude (0.003–149 g NO_3^--N m^{-2} $year^{-1}$) (e.g., Hernandez and Mitsch 2007; Mitsch and others 2005; Teiter and Mander 2005). Nitrate concentration reductions are also extremely variable, with some treatment wetlands leading to increases in nitrate concentrations and others removing close to 100 % of nitrate entering the wetland (Fisher and Acreman 2004; Kadlec 1994; Nahlik and Mitsch 2006). Generally, treatment wetlands with low or negative nitrate-removal rates are well-aerated (nonsaturated) wetlands where organic C mineralization and nitrification can lead to increases in nitrate concentration at the outlet. Wetlands with organic rich, permanently saturated soils generally demonstrate high nitrate removal rates (Mitsch and Gosselink 2000). In a review of regional and global effects of wetlands, Verhoeven and others (2006) estimated that significant improvement of water quality by removing N could be obtained if at least 2–7 % of the catchment area consisted of wetlands.

Compared with natural wetlands, constructed wetlands generally have lower N-removal efficiencies. For instance, Hammer and Knight (1994) reported N-removal reductions averaging 44 % in 17 constructed wetlands compared with 77 % in 26 natural wetlands. Forty percent N reduction was reported in a series of stormwater wetlands in North Carolina (USA) (North Carolina Department of Environment and Natural Resources 2005), whereas the Best Management Practice (BMP) database reports a 62 % decrease for constructed stormwater wetland basins (BMP database 2010) (Table 1). Generally, the greater efficiency of natural wetlands is associated with the denser vegetation often observed in these systems compared with treatment wetlands (Bastviken and others 2009; Kadlec 2005; Tanner and others 2005). Vegetation in natural wetlands is also often more mature, and organic C concentration is usually greater than in constructed treatment wetlands (Appelboom and Fouss 2006; Craft 1997). Over time, C availability in constructed treatment wetlands will increase owing to vegetation decay and will help support soil denitrification (Reddy and Patrick 1984). Planting mixed vegetation in constructed wetlands may also help increase N removal by way of denitrification because mixed vegetation tends to promote greater denitrification rates compared with single-species stands (Bachand and Horne 2000). Wetland soil composition and structure also influence treatment effectiveness. For instance, a wetland substrate with a C:N ratio ≥5:1 will prevent C limitation in most cases (Baker 1998). In addition, soil particle size will impact water residence time, the development of anoxic conditions, and the flow of water, which in turn will affect N removal.

When the primary goal of installing treatment wetlands is N removal, preference should be given to HSSF wetland types over VSSF and FSW wetland types because HSSF wetland systems are generally permanently saturated, which promotes the development of decreasing conditions favorable to N removal by way of heterotrophic denitrification (Kadlec and Wallace 2008). However, high concentrations of suspended solids in the influent can limit flow in the subsurface and ultimately decrease N removal (Hammer 1992). Regardless of type, treatment wetlands are generally most efficient when located at the outlet of small drainage basins where N concentrations are less diluted than in larger watersheds (Mitsch 1992).

The capital cost for installing a treatment wetland is highly dependent on the size of the wetland (or series of wetlands), configuration (horizontal subsurface flow wetland, FWS wetland), and regional market costs (Kadlec and Wallace 2008). In the US, capital costs range from \$0.001 to \$0.1/m^2 (\$10–\$1,000/ha) for FWS wetlands and from \$0.03 to \$1/m^2 (\$300–\$10,000/ha) for HSSF or VSSF wetlands. The latter are generally more expensive per unit area due to the cost of gravel. However, HSSF wetlands are typically much smaller than FWS wetlands. After initial installation, constructed wetlands generally require few operational costs other than the occasional removal of sediments. However, management of vegetation (e.g., plant harvesting), mosquitoes (e.g., insecticide spraying), and animals (e.g., repairing damage caused by muskrats, geese, etc.) may generate additional costs. Overall, treatment wetlands are among the most cost-effective systems for treating large volumes of N-contaminated waters. Preference should be given to HSSF wetlands for N removal in urban or industrial areas (because of greater N-removal efficiencies), whereas FWS wetlands should be the preferred option in agricultural areas where large sediment loads could quickly decrease N-removal efficiency in the more engineered HSSF and VSSF wetland types.

Approach No. 5: Managed Riparian Buffers

Riparian zones (i.e., vegetation adjacent to water bodies) often naturally occur in the landscape. Due to their potential nutrient removal benefits and other ecosystem services (Palone and Todd 1997; Dosskey 2001; Hill 1996; Puckett 2004), many are intensively managed and/or engineered (e.g., as part of large stream-restoration projects) and are widely recommended as best-management practices by federal and local agencies around the world (Lowrance and others 1997; Naiman and others 2005; Welsch 1991) (Fig. 9). Recommendations for effectively managing nitrogen in riparian buffers often rely on a three-tiered approach for optimal nutrient and sediment buffering capacity (Schultz and others 1995). First, a region of undisturbed forest near the stream should be maintained to ensure stream bank stability and limit erosion. Moving further away from the stream, an area of managed forest can be established to improve uptake of N transported in deeper groundwater flow paths (Schulz and others 1995; Lowrance and others 1997). Adjacent to the upland, a grassy area should be maintained to promote infiltration and trap any contaminants present in overland flow.

Because nitrate assimilation and uptake by vegetation are only temporary storage mechanisms, heterotrophic

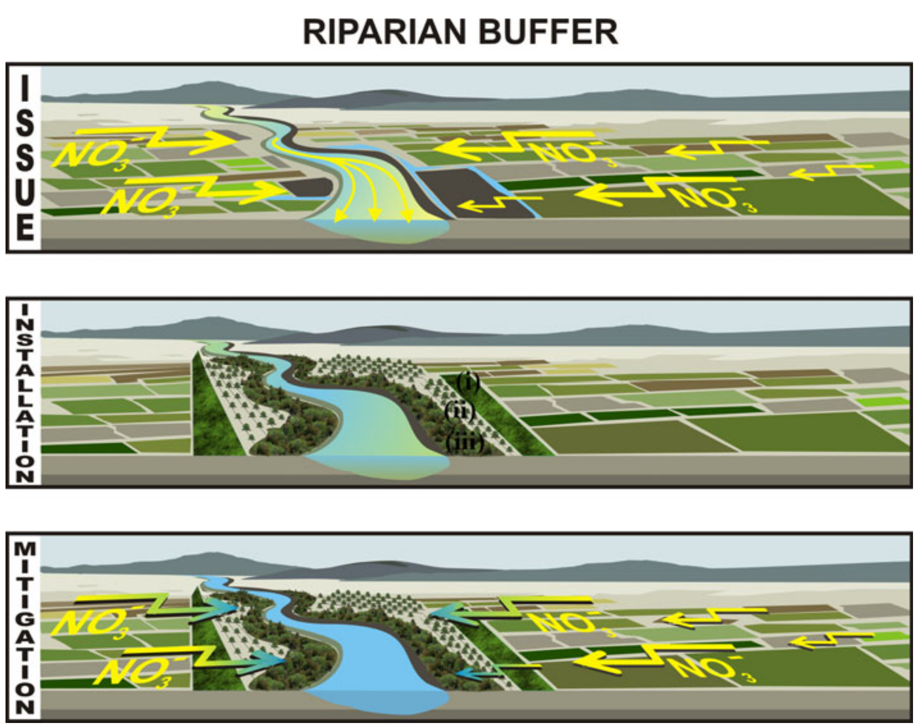

Fig. 9 Riparian buffer. (*Issue*) Agricultural N derived from fertilizer applications flows toward receiving rivers. (*Installation*) Three tier riparian buffer zone: grassed area or runoff control zone (i), managed forest (ii), and undisturbed forest (iii) (after Lowrance and others 1997). Nonpoint source N is intercepted by the buffer before it reaches a stream or receiving water body. The three tiers are intended to be effective at intercepting various surface and shallow to deep subsurface flow paths. (Mitigation) N may be assimilated by vegetation or consumed by heterotrophic denitrifying bacteria in the soil

denitrification is generally considered the most important N-removal process in riparian zones (Vidon and others 2010). However, fast-growing trees (e.g., willows, poplars, etc.), when harvested on a regular basis, can provide timber and significantly contribute to N removal from the subsurface, especially during the growing season (Newbold and others 2010). Tree harvesting has the potential to negatively affect the other ecosystem services provided by riparian zones, such as habitat for wildlife, recreation, or bank stabilization. Therefore, multiple benefits must be weighed before harvesting is considered as an N-management approach.

A 90 % decrease in nitrate concentration in subsurface flow in the riparian zone is generally achieved ≤20 m from the field edge unless riparian sediments are coarse sand and/or gravel, in which case a 50-m width is generally required (Hill 1996; Gold and others 2001; Burt and others 2002; Vidon and Hill 2006; Zhang and others 2010). A recent study in Alberta, Canada, suggested that width be calculated based on surficial geology (20 m in glacial till landscapes, 50 m in alluvial and outwash landscapes) with modifiers based on slope (Alberta Environment 2012). Specifically, this study recommended that the managed riparian zone be widened by 1.5 m for every 1 % slope >5 % (Alberta Environment 2012). A recent review of the literature also showed that statistically, N removal in riparian zones tends to increase with riparian width. Mayer and others (2007) indicate that N-removal efficiency of riparian zone width categories 0–25, 26–50, and >50 m were approximately 58, 71, and 85 %, respectively. Buffers composed of trees also tend to have greater N-removal efficiencies than buffers composed of grasses or mixtures of grasses and trees (Zhang and others 2010). Regardless of vegetation, subsurface hydrology (saturated *vs*. unsaturated soil conditions) and redox condition appear to be significant determinants of N-removal efficiency (Mayer and others 2007, 2010). Despite high N removal observed in most managed riparian zones, several studies have documented potential or actual nitrate leaching in aerobic soils in urban riparian buffers located next to incised streams or in landscapes with regional lowering of groundwater tables due to decreased infiltration caused by impervious cover (Groffman and others 2002; Stander and Ehrenfeld 2009). Thus, hydrologic connectivity between the managed riparian buffer and the stream is a critical factor to ensuring efficient N removal in buffer zones.

Costs of using riparian zones to decrease N delivery to streams depend on the situation and are often not easily available. If the riparian zone is vegetated and hydrologically connected between the upland and stream, there may be no cost at all, other than the cost and effort of negotiating an easement with the landowner to ensure continued integrity of the buffer. For instance, in Pennsylvania, securing easements to preserving farmlands from development and urbanization and, consequently, to preserve their associated riparian areas, has an estimated one-time cost of \$7,400/ha on average, to which approximately \$10,000/project are needed for transactional work (e.g., survey, recording fees, staff time) (Jeffery E. Swinehart, personal communication). However, where riparian buffers do not exist and/or must be revegetated or rebuilt (e.g., by way of stream bank reengineering), costs can increase quickly. For instance, Roberts and others (2009) estimated the annual costs of establishing and maintaining a 45.7-m (150-foot) riparian buffer adjacent to agricultural land within the Harpeth River watershed in Tennessee (USA) to be approximately \$0.0262/m^2 [\$262/ha (\$0.0024/square foot)] of riparian buffer/year. Expenses may involve vegetation management (tree harvesting, removal of invasive species) or cost associated with taking agricultural land out of production. Often, it is difficult to determine with precision the actual cost of riparian zone installation from published data because riparian zone-restoration generally occurs within the framework of larger stream-restoration projects where riparian zone costs are included in the cost of the entire stream-restoration project.

Ultimately, managed riparian buffers have the potential to significantly decrease N non–point source pollution provided the following conditions are met: (1) riparian zone placement should allow for efficient runoff interception and significant interactions between N-laden subsurface flow and organic rich surficial riparian soils (Dosskey and Qiu 2010); (2) vegetation cover should be adequate and species composition diverse enough, including trees, to decrease erosion and help maintain the soil organic C content during long periods of time (e.g., decades) (Dosskey and others 2010); and (3) riparian zones should be wide enough to remove most N in the subsurface. Wider buffers are generally more effective at attenuating nitrogen, but width should be adjusted to local conditions (soil texture, slope) (Alberta Environment 2012).

Approach No. 6: Artificial Lakes and Reservoirs

Although many states and municipalities have recently removed small dams and/or reservoirs (e.g., old mill dams) for various social, economic, and ecological reasons (Doyle and others 2008; Orr and others 2004), artificial lakes and reservoirs remain ubiquitous structures in the landscapes (Graf 1999) (Fig. 10). The >84,000 dams across the United States and their associated artificial lakes and reservoirs are primarily designed for water storage, flood control, hydropower, and recreation (United States Army Corps of Engineers 2011), but many can serve as significant N sinks (David and others 2006; Harrison and

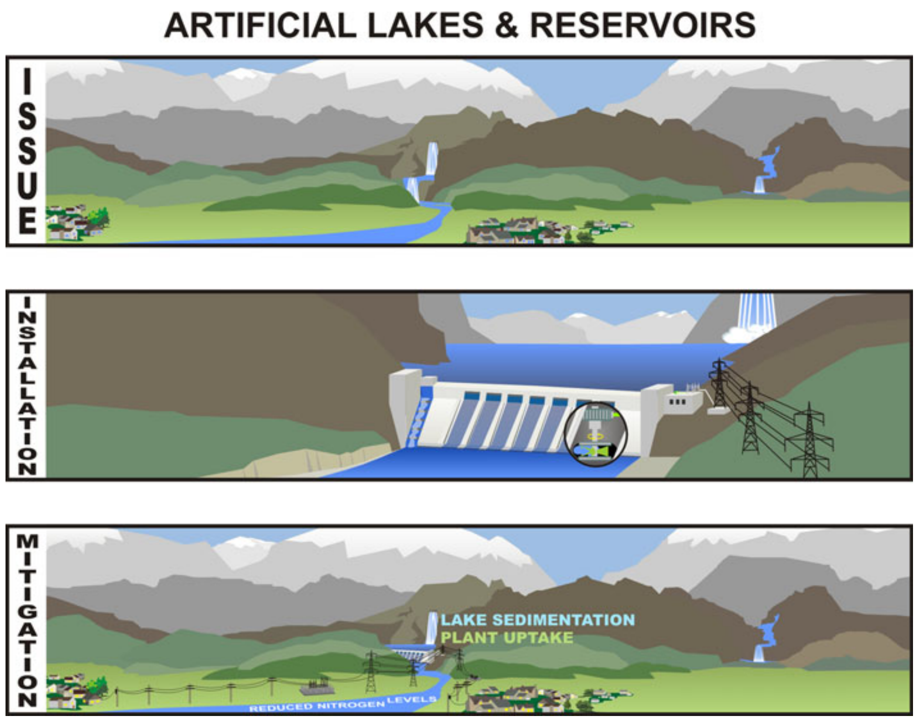

Fig. 10 Artificial lakes and reservoirs. (*Issue and Installation*) Artificial lakes and reservoirs are constructed for various purposes, such as flood control, hydroelectric power generation, or water storage. (*Mitigation*) N may be assimilated by algae and vegetation growing in lakes and reservoirs, buried in deep sediments, or removed by microbial activity in the sediments

others 2009; Seitzinger and others 2006). The efficiency and potential of these artificial lakes and reservoirs to attenuate N varies widely depending on key controls, including concentration and timing of the total N load entering the reservoir (Gruca-Rokosz and others 2009; Wall and others 2005), physical placement within the watershed (Kellogg and others 2010), and hydraulic residence time (Seitzinger and others 2006).

Mechanisms responsible for N removal in reservoirs include denitrification, sedimentation that can lead to burial of N-containing particles, and biological uptake by plants and microbes. The temporal regime of N delivery to a reservoir is also critical to the efficiency of N retention, particularly during cold periods that limit microbial denitrification (Braskerud 2002; Wall and others 2005). Larger reservoirs with greater residence times may be less sensitive to seasonal temperature and hydrologic flushing effects compared with other habitats, such as wetlands (Jansson and others 1994). When large plant or algal communities are present, biological uptake can dominate retentive processes, especially during high growth periods (e.g., summer). Although biological uptake only temporarily removes N, biomass (plants and algae) may be harvested to permanently remove N (Carpenter and Adams 1977; Hill 1979). Burial in a reservoir is likely to be slow and dependent on deposition and ammonium mineralization rates because inorganic N rapidly cycles through several chemical forms that are highly mobile. Denitrification generally occurs in anoxic interstices of benthic and littoral sediments (Christensen and Sorensen 1986; Seitzinger 1988; Saunders and Kalff 2001) and, to a lesser degree, in the anoxic hypolimnia of reservoirs (Seitzinger 1988).

Most studies that measure N-removal efficiency in artificial lakes and reservoirs indicate that these systems are generally significant N sinks but that specific N-removal efficiencies vary widely. In a review of N-removal data in reservoirs, Kellogg and others (2010) found that N loss was positively related to hydraulic residence time, with overall N removal varying between 10 and 100 % (Table 1). Other studies report N-removal efficiencies varying from 3 to 20 % depending on the location (Braskerud 2002; Deemer and others 2011). In an urban pond, Rosenzweig and others (2011) reported that N removal varied from −10 % (N production) to 68 %, with season and temperature acting as primary controls. Artificial lakes and reservoirs, especially large systems associated with long residence times, therefore generally act as N sinks. However, seasonal N-removal dependence may be observed in small reservoirs where denitrification and plant uptake can significantly increase during warmer months.

Although existing artificial lakes and reservoirs generally act as N sinks and could therefore be seen as components of any N-management plan when already present in a watershed, the construction of new reservoirs or artificial lakes is not recommended for N management. Indeed, the negative impact of reservoirs on river hydrology and ecosystems often outweigh water-quality benefits with respect to N. For instance, reservoir construction may cause significant losses of N-removal hot spots in floodplains (Forshay and Stanley 2005), streams (Forshay and

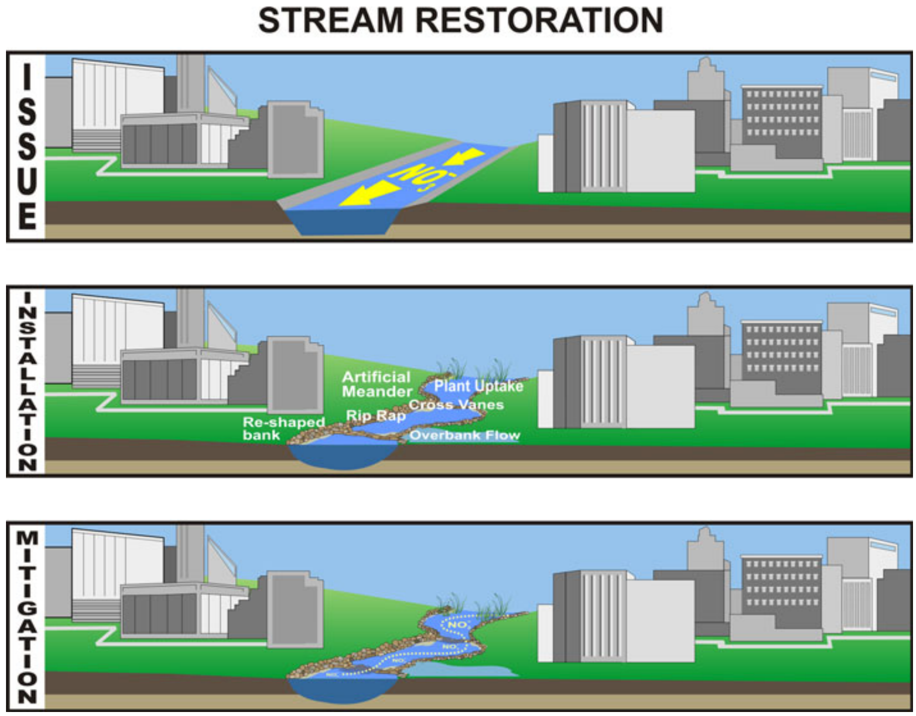

Fig. 11 Stream restoration. (*Issue*) Concrete straight channel may rapidly convey N to rivers. (*Installation*) Illustration of various stream-restoration techniques, such as reshaped banks, cross vanes, artificial meander, and rip-rap, implemented in a conceptualized urban watershed. Most techniques are designed to stabilize stream banks and decrease erosion. These same techniques may also increase groundwater residence time, reconnect floodplains to stream channels, and enhance plant growth. (*Mitigation*) N uptake by algae and plants may be enhanced, and denitrification may increase in response to changes in hydrology and availability of organic C

Dodson 2011), and managed riparian buffers (Mayer and others 2007) because of associated habitat loss and changes to the hydrology that governs N removal in these habitats. Furthermore, environmental factors, such as the obstruction of fish migration routes (Opperman and others 2009) and long-term maintenance and dredging costs (Doyle and others 2008), are critical considerations in deciding to build new artificial lakes and reservoirs for N management. Consequently, the addition of new large artificial lakes and reservoirs may only be recommended for N management in cases where water storage, flood control, hydropower, and recreation are the primary needs and where N removal is an added benefit. In the rare cases where the construction of new artificial lakes and reservoirs should be recommended, construction costs vary from several thousand dollars for a small farm pond to tens of millions of dollars for hydropower systems. In these cases, a network of small, shallow artificial lakes and reservoirs to maximize surface water area, wet littoral zones, water residence time, and organic C loading will generally yield greater N removal than a single large reservoir.

Approach No. 7: Stream Restoration

Stream restoration runs a gamut of techniques and objectives, and rates of N removal are variable based on hydraulic residence times and position along stream networks (Kaushal and others 2008a; Sivirichi and others 2011). Craig and others (2008) categorized stream-restoration techniques into five broad overlapping categories: (1) organic matter additions (e.g., artificial debris dams, woody debris); (2) channel reconstruction (channel widening, weirs, and cross vanes); (3) floodplain reconnection (wetland benches, bank grading, and reshaping); (4) artificial geomorphic features (oxbows, side channels, ponds); and (5) bank stabilization (rip-rap, erosion cloth, root wads) (Fig. 11). Often, urban stream-restoration projects target buried streams (*sensu* Elmore and Kaushal 2008), in which channels are encased in concrete or pipes (Duerksen and Snyder 2005), and stream reaches affected by sanitary sewer leaks (Sivirichi and others 2011), where restoration is intended to address numerous environmental impacts. In a national survey, general water-quality improvement was the most frequently stated restoration goal (approximately 30 % of the time), but only recently has N removal been a primary goal for stream restoration (Bernhardt and others 2005). Often, restoration is guided by natural channel design (NCD), a suite of techniques based on a stream classification system and fluvial geomorphologic principles (Rosgen 1994, 1996). The NCD approach has been controversial for its rigidity and alleged lack of supporting evidence for its effectiveness (Bernhardt and Palmer 2011; Lave 2009). Thus, there remains no comprehensive, universally accepted reference to follow for restoring streams (Federal Interagency Stream Restoration Working Group 1998), and much effort is currently being devoted to identify information gaps (Wenger and others 2009) and establish criteria for stream-restoration effectiveness (Bernhardt and Palmer 2007; Palmer and Filoso 2009).

Recent studies of restored stream performance on N reduction quantified the capacity of natural or constructed features to enhance N uptake, including implanted logs, riffle and step structures that increase hyporheic exchange (Bukaveckas 2007; Kasahara and Hill 2006a, b), debris dams that retain C used by denitrifiers (Groffman and others 2005), and streams where C supplies are increased by plants (Gift and others 2010). Other studies have investigated designs that could create hot spots or "hot moments" of N removal by hydrologically reconnecting the channel to the floodplain (Fink and Mitsch 2007; Harrison and others 2011; Kaushal and others 2008b; Opperman and others 2009). These include designs that allow hyporheic exchange and overbank flow during significant precipitation events (Kaushal and others 2008b), pond-and-oxbow features that divert water from the main channel into highly biologically active wetlands (Fink and Mitsch 2007; Harrison and others 2011), and large-scale hydrologic reconnection to river floodplains (Opperman and others 2009).

Stream restoration can improve N processing, particularly in urban streams, if the restoration approach incorporates mechanisms that slow down stream flow, increase hydraulic residence time, increase availability of dissolved organic C, and/or hydrologically reconnect the stream channel to floodplain wetlands and riparian zones (Bukaveckas 2007; Filoso and Palmer 2011; Gift and others 2010; Groffman and others 2005; Harrison and others 2011; Kaushal and others 2008b; Klocker and others 2009; Roberts and others 2007; Sivirichi and others 2011). Other design considerations include adapting stream-restoration strategies to land use and the timing and intensity of N export (Filoso and Palmer 2011; Shields and others 2008). For instance, low-density suburban catchments export total N and nitrate loads mostly at relatively low flows, whereas more urbanized sites export total N and nitrate at higher and less frequent flows (Shields and others 2008). In urban catchments, N retention may be limited during high flows (Kaushal and others 2011). Therefore, stream restoration will be most effective at managing N if approaches include methods to decrease stream flashiness and increase groundwater residence time (Craig and others 2008; Kaushal and others 2008a; Mayer and others 2010) and/or if stream restoration is used in conjunction with other EEPs that improve stream bank stability and increase water retention during storms (Selvakumar and others 2010). Stream restoration may be most effectively employed in areas of low-density development served by septic systems where N loads are consistent and systems less flashy (Shields and others 2008).

Some studies have suggested that there may be little or no effect of stream restoration on N-uptake rates and that stream restoration contributes to tree removal in riparian zones during the construction phase (Sudduth and others 2011). It is often difficult to compare N removal between stream-restoration projects because metrics for N-removal rates are inconsistent and differ across varying spatial and temporal scales of monitoring. Nevertheless, N-removal data recently available from studies in the Maryland coastal plain (USA) show that restoration efforts that created stream–wetland complexes decreased N during storm flow and that overall N removal was approximately 5 % of inputs (Filoso and Palmer 2011). Other work in the Maryland Piedmont region has shown that stream–wetland complexes can remove approximately 10–40 % of N depending on hydraulic residence time (Harrison and others 2011; Kaushal and others 2008a; Klocker and others 2009; Sivirichi and others 2011). Further studies are necessary to characterize stream-restoration effectiveness at the larger stream network scale (Sivirichi and others 2011), and adequately assess the importance of groundwater and surface water interactions (Mayer and others 2010). Alternative approaches to NCD designed specifically to reconnect groundwater to surface water (Parola and Hansen 2011) are being employed to restore streams impacted by legacy sediments deposited during mill pond construction and colonial era agricultural erosion (Walter and Merritts 2008). The efficacy of such approaches for decreasing N is currently under study (Hartranft and others 2011).

Costs associated with stream-restoration vary widely, with median costs ranging from $15,000 to $812,000/project and a median cost of $19,000 for projects specifically targeted toward water-quality management (Bernhardt and others 2005). Three stream-reach scale projects in Baltimore County, MD, USA, that incorporated extensive stream channel restructuring and installation of hard engineered structures, such as cross vanes, rock weirs, and oxbow ponds (Harrison and others 2011), ranged in cost from $520 to $1526/m, including the costs of new bridges and road infrastructure (USEPA 2006). The objectives of these projects were not limited to nutrient control but also included erosion control, protection of sewer infrastructure, and fish passage. Collectively 26 stream-restoration projects in Baltimore County, MD covering approximately 16,090 m of streams cost $12.4 million, an average of $770/m (Duerksen and Snyder 2005). Restoration of Big Spring Run, a stream near Lancaster, PA, USA, involving removal of legacy sediments and the construction of a multichannel, stream-wetland complex within a 4.35 km^2 watershed cost $600,000 and resulted in the restoration of 915 m of stream, an average of $655/m (J. Hartranft, personal communication, June 13, 2012).

Recommendations and Future Research Needs

Selection of the appropriate EEPs for N management depends on N source, hydrology, land use, availability of

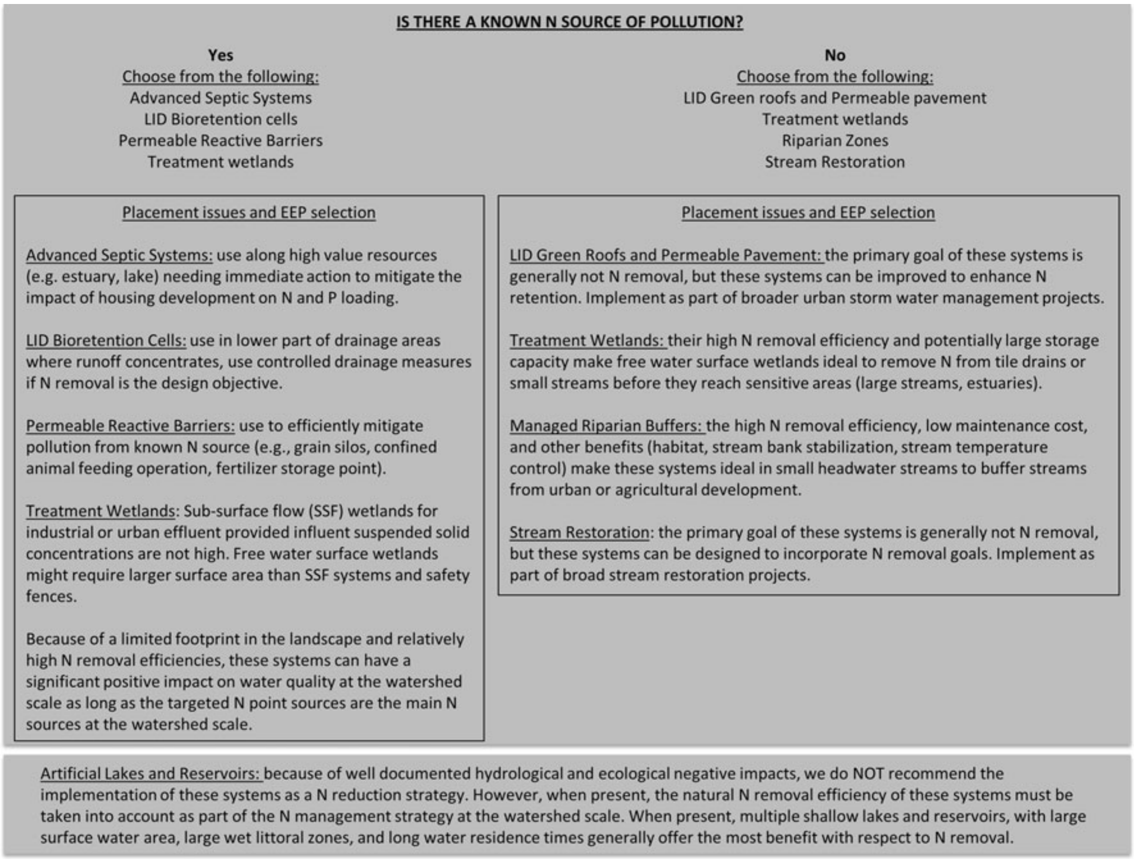

Fig. 12 Decision-making template for EEP implementation and selection for N-pollution mitigation in actively managed watersheds

land for EEP implementation, available budget, and ancillary management objectives. Figure 12 summarizes the pros and cons of the seven EEPs discussed in this review. This figure is intended to help landscape managers make more informed decisions about when and where to implement one or a combination of EEPs based on the N source, the existence of artificial lakes and reservoirs, and overall management goals. The identification and characterization of N loads and of the hydrology of the contributing area (continuous N pollution *vs*. flashy N load events) is necessary to chose among techniques. Indeed, many EEPs with high N-removal efficiencies may be overwhelmed during high-flow periods (e.g., managed riparian zones, PRBs), or are known to function better at low flow than during flashy storm events (e.g., stream restoration). In addition, land-use characterization tools must be used to locate potential natural N sinks (e.g., vegetated riparian areas, wetlands, lakes) and ensure that watershed management plans will not disconnect natural hydrology from N flows. For example, in agricultural settings with both non-point sources of N (e.g., fertilizer application on crops) and point sources of N (e.g., grain silos, confined animal feeding operations), employing managed riparian buffers throughout the watershed, along with the targeted use of PRBs for N removal at select locations of point source N, may be an efficient way of combining EEPs. In either urban or agricultural landscapes, the use of treatment wetlands at select locations receiving large amounts of N-rich runoff might provide both peak flow mitigation and high N removal and could be used upstream of stream-restoration projects. Similarly, advanced septic systems for houses located along sensitive areas (e.g., estuary, lakes) might provide added benefits if used in concert with effective protection measures for streams in the watershed (e.g., managed riparian zones, stream restoration) and with LID structures in urbanized areas of the watershed. Ultimately, a combination of EEPs working together to decrease peak flow, intercept point sources of N before they reach a stream, protect streams from direct N contamination, and/or enhance nutrient processing in streams will likely be more efficient at removing N than any of the EEPs presented here used alone.

Recent developments in geospatial techniques (geographic information systems, LiDAR, digital elevation models), and broadly available digital databases containing elevation data (National Elevation Data set, United States Geological Survey [USGS] 2006), vegetation and land cover (National Landcover Data set USGS 2011a), soil and geomorphology data (Natural Resources Conservation Service 2011), and/or hydrological data (National Hydrography Data set, USGS 2011b), offer improved opportunities for landscape managers and scientists to engage in scenario modeling (e.g., Kellogg and others 2010) and better define how EEPs can be used in the landscape to optimize N removal while minimizing costs. For instance, recent studies have engaged in scenario-modeling to optimize EEP placement in landscapes for maximizing N-removal benefits at the watershed scale (Dosskey and Qiu 2010; Kellogg and others 2010). However, scenario-modeling efforts are currently hindered by a lack of summary information on the suitability of various EEPs to mitigate N pollution in a variety of settings.

Overall, our analysis identified four major areas where more knowledge or more integration is critically needed to better predict N-removal potential of one or several EEPs. First, much more interaction between engineers working on "hard-engineering structures" (e.g., LID bioretention cells) and scientists/managers using "soft-engineering approaches" (e.g., managed riparian zones, stream restoration) is needed to fully assess how to best use various approaches in concert, at the watershed scale, to achieve water-quality goals. For instance, it is likely that the development of LID structures at headwater locations could lead to decreased peak flow during storms. Decreased peak flow could help optimize stream-restoration efficiency of N removal because stream-restoration structures are generally more efficient at removing N during relatively low-flow compared with high-flow conditions.

Second, there is a need to develop a set of homogenous metrics across disciplines to assess cost and N-removal efficiencies. Often, mass removal is more important than the percent removal itself in identifying effective EEPs. Currently, some studies report N mass removal in mass removed per volume of soil, per square meter, or per meter of stream length. Converting these units into a single set of unit of N mass removed would require making many assumptions about EEP size, contributing area, residence time, soil porosity, etc. Such information should be provided in future studies. In addition, we recommend that further studies report total inorganic nitrogen because nitrate and ammonium are often the primary forms of N associated with negative ecosystem impacts, such as eutrophication. When possible, influent and effluent N masses and water volumes should be provided for a given EEP. Cost data are also reported in a variety of units, including cost per impervious acre treated, per acre of drainage area, per square foot of the practice, or per cubic foot of runoff treated (Cappiella and Hirschman 2012). Some studies also report full life-cycle costs (e.g., design, construction, maintenance), whereas others only report construction costs. The lack of homogenous metrics for cost and N-removal efficiency makes it difficult to compare EEPs. Useful metrics would include cost per unit of N removed or influent N load.

Third, there is a need for more research on the long-term efficiency and key processes regulating the functioning of EEPs in a variety of climatic and physiographic regions so that variability in N-removal rates can be better evaluated. For instance, many studies have reported how climatic and landscape geomorphic characteristics impact N removal in riparian zones (Gold and others 2001; Sabater and others 2003; Vidon and Hill 2006), but few have reported how these important variables impact N removal in LID systems, PRBs, or streams. Furthermore, many studies reporting N-removal efficiencies rely on only a few data points, which hinders our ability to fully assess the significance of reported N-removal rates.

Finally, as efforts are made to better understand where and when to place EEPs in watersheds to optimize N removal, there is also a critical need to better quantify and value, both economically and in terms of ecosystem services, the environmental tradeoffs associated with each of the practices discussed here. For instance, artificial lakes and reservoirs often contribute to the disconnection of the river to its floodplain, block fish migration routes, and alter natural flow regimes important for some biota (Forshay and Dodson 2011; Forshay and Stanley 2005; Opperman and others 2009). Some managed riparian zones can be significant sources of P to streams, and some wetlands can contribute to the release of methylmercury in the environment (Carlyle and Hill 2001; Mitchell and others 2006, 2008). Nevertheless, EEPs are a cost-effective approach to managing excess N in human-influenced landscapes, especially where N-source control is not possible. Some EEPs may have additional value, such as providing green space or wildlife habitat, and therefore the costs of implementing EEPs can be spread among multiple, stacked benefits.

Acknowledgments The authors thank George Loomis, research and extension soil scientist and director of the New England Onsite Wastewater Training Center, for sharing his expertise in advanced-treatment septic systems and Kathy Tynsky for creating the figures. This paper is a product of a workshop, Managing Denitrification in Human Dominated Landscapes," held May 12–14, 2009, at the University of Rhode Island Bay Campus, Narragansett, RI, USA, with support from the Denitrification Research Coordination Network of the National Science Foundation (award DEB0443439) and the United States Department of Agriculture CSREES Northeast States. Stream restoration research was funded by Maryland Sea Grant Awards SA7528085-U, NA05OAR4171042, and R/WS-2, and by the United States Environmental Protection Agency through its Office of Research and Development under cooperative agreement CR829676. The research described in this article has not been subjected to USEPA review and therefore does not necessarily reflect the views of any of the funding agencies, and no official endorsement should be inferred.

References

Alberta Environment (2012) Stepping back from the water: A beneficial management practices guide for new development near water bodies in Alberta's settled region. AE, Calgary. http://environment.gov.ab.ca/info/library/8554.pdf. Accessed 16 Nov 2012

Albuquerque A, Oliveira J, Semitela S, Amaral L (2009) Influence of bed media characteristics on ammonia and nitrate removal in shallow horizontal subsurface flow constructed wetlands. Bioresource Tech 100:6269–6277

Appelboom T, Fouss J (2006) Methods for removing nitrate N from agricultural drainage waters: a review and assessment. In: Proceedings of the American Society of Agricultural and Biological Engineers International (ASABE), St. Joseph

Bachand PAM, Horne AJ (2000) Denitrification in constructed free-water surface wetlands: II. Effects of vegetation and temperature. Ecol Eng 14:17–32

Baker LA (1998) Design considerations and applications for wetland treatment of high-nitrate waters. Water Sci Technol 38:389–395

Bastviken SK, Weisner SEB, Thiere G, Svensson JM, Ehde PM, Tonderski KS (2009) Effects of vegetation and hydraulic load on seasonal nitrate removal in treatment wetlands. Ecol Eng 35:946–952

Bernhardt ES, Palmer MA (2007) Restoring streams in an urbanizing world. Freshwater Biol 52:738–751

Bernhardt ES, Palmer MA (2011) River restoration: The fuzzy logic of repairing reaches to reverse catchment scale degradation. Ecol Appl 21:1926–1931

Bernhardt ES, Palmer MA, Allan JD, Alexander G, Barnas K, Brooks S et al (2005) Synthesizing U.S. river restoration efforts. Science 308:636–637

BMP database (2010) International stormwater Best Management Practices (BMP) database pollutant category summary: Nutrients. Prepared by Geosyntec Consultants, Inc., and Wright Water Engineers, Inc. http://www.bmpdatabase.org/Docs/BMP%20Database%20Nutrients%20Paper%20December%202010%20Final.pdf. Accessed: August 13, 2012

Braskerud BC (2002) Factors affecting nitrogen retention in small constructed wetlands treating agricultural non-point source pollution. Ecol Eng 18:351–370

Bukaveckas PA (2007) Effects of channel restoration on water velocity, transient storage, and nutrient uptake in a channelized stream. Environ Sci Technol 41:1570–1576

Burgin AJ, Hamilton SK (2008) NO_3^- driven SO_4^{2-} production in freshwater ecosystems: Implications for N and S cycling. Ecosystems 11:908–922

Burt TP, Pinay G, Matheson FE, Haycock NE, Butturini A, Clement J-C et al (2002) Water table fluctuations in the riparian zone: Comparative results from a pan-European experiment. J Hydrol 265:129–148

Cappiella K, Hirschman D (2012) Runoff ramblings: "Cost-effective practices:" The devil is in the details. Runoff Rundown, issue no. 46. http://www.cwp.org/newsroom/runoff-rundown.html. Accessed: April 27, 2012

Carlyle GC, Hill AR (2001) Groundwater phosphate dynamics in a river riparian zone: Effects of hydrologic flowpaths, lithology, and redox chemistry. J Hydrol 247:151–168

Carpenter SR, Adams MS (1977) The macrophyte tissue nutrient pool of a hardwater eutrophic lake: Implications for macrophyte harvesting. Aquat Bot 3:239–255

Carpenter SR, Caraco NF, Correll DL, Howarth RW, Sharpley AN, Smith VH (1998) Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecol Appl 8:559–568

Cho KW, Song KG, Cho JW, Kim TG, Ahn KH (2009) Removal of nitrogen by a layered soil infiltration system during intermittent storm events. Chemosphere 76:690–696

Christensen P, Sorensen J (1986) Temporal variation of denitrification activity in plant-covered, littoral sediment from Lake Hampen, Denmark. Appl Environ Microbiol 51:11–74

Collins KA, Hunt WF, Hathaway JM (2008) Hydrologic comparison of four types of permeable pavement and standard asphalt in eastern North Carolina. J Hydrol Eng 13:1146–1157

Collins KA, Lawrence TJ, Stander EK, Jontos RJ, Kaushal SS, Newcomer TA et al (2010a) Opportunities and challenges for managing nitrogen in urban stormwater: A review and synthesis. Ecol Eng 36:1507–1519

Collins KA, Hunt WF, Hathaway JM (2010b) Types of permeable pavement and standard asphalt in eastern North Carolina. J Hydrol Eng 15:512–521

Craft CB (1997) Dynamics of nitrogen and phosphorus retention during wetland ecosystem succession. Wetl Ecol Manag 4: 177–187

Craig LS, Palmer MA, Richardson DC, Filoso S, Bernhardt ES, Bledsoe BP et al (2008) Stream restoration strategies for reducing river nitrogen loads. Front Ecol Environ 6:529–538

Czemiel Berndtsson J, Emilsson T, Bengtsson L (2006) The influence of extensive vegetated roofs on runoff water quality. Sci Total Environ 355:48–63

Czemiel Berndtsson J, Bengtsson L, Jinno K (2009) Runoff water quality from intensive and extensive vegetated roofs. Ecol Eng 35:369–380

David MB, Wall LG, Royer TV, Tank JL (2006) Denitrification and the nitrogen budget of a reservoir in an agricultural landscape. Ecol Appl 16:2177–2190

Deemer BR, Harrison JA, Whitling EW (2011) Microbial dinitrogen and nitrous oxide production in a small eutrophic reservoir: An in situ approach to quantifying hypolimnetic process rates. Limnol Oceanogr 56:1189–1199

Diaz RJ, Rosenberg R (2008) Spreading dead zones and consequences for marine ecosystems. Science 321:926–928

Dietz ME (2007) Low impact development practices: A review of current research and recommendations for future directions. Water Air Soil Pollut 186:351–363

Dietz ME, Clausen JC (2006) Saturation to improve pollutant retention in a rain garden. Environ Sci Technol 40:1335–1340

Dosskey MG (2001) Toward quantifying water pollution abatement in response to installing buffers on crop land. Environ Manage 28:577–598

Dosskey MG, Qiu Z (2010) A comparison of alternative methods for prioritizing buffer placement in agricultural watersheds for water quality improvement. In: Proceedings of the 10th International Conference on Precision Agriculture, Denver, CO, July 18–21

Dosskey M, Vidon P, Gurwick NP, Allan CJ, Duval T, Lowrance R (2010) The role of riparian vegetation in protecting and improving chemical water quality in streams. J Am Water Resour Assoc 46:261–277

Doyle MW, Stanley EH, Havlick DG, Kaiser MJ, Steinbach G, Graf WL et al (2008) Aging infrastructure and ecosystem restoration. Science 319:286–287

Duerksen C, Snyder C (2005) Nature-friendly communities: habitat protection and land use planning. Island Press, Washington

Elmore AJ, Kaushal SS (2008) Disappearing headwaters: patterns of stream burial due to urbanization. Front Ecol Environ 6:308–312

Federal Interagency Stream Restoration Working Group (1998) Stream corridor restoration: principles, processes, and practices. Government Printing Office Item No. 0120-A, SuDocs No. A 57.6/2:EN 3/PT.653

Filoso S, Palmer MA (2011) Assessing stream restoration effectiveness at reducing nitrogen export to downstream waters. Ecol Appl 21:1989–2006

Fink DF, Mitsch WJ (2007) Hydrology and nutrient biogeochemistry in a created river diversion oxbow wetland. Ecol Eng 30:93–102

Fisher J, Acreman MC (2004) Wetland nutrient removal: a review of the evidence. Hydrol Earth Syst Sci 8:673–685

Forshay K, Dodson S (2011) Macrophyte presence is an indicator of enhanced denitrification and nitrification in sediments of a temperate restored agricultural stream. Hydrobiologia 668:21–34

Forshay KJ, Stanley EH (2005) Rapid nitrate loss and denitrification in a temperate river floodplain. Biogeochemistry 75:43–64

Gift D, Groffman PM, Kaushal SS, Mayer PM, Striz EA (2010) Root biomass, organic matter and denitrification potential in degraded and restored urban riparian zones. Restor Ecol 18:113–120

Gold AJ, Sims JT (2000) Research needs in decentralized wastewater treatment and management: a risk-based approach to nutrient contamination. In: National research needs conference draft proceedings: risk-based decision making for onsite wastewater treatment. St. Louis, MO. http://www.uri.edu/cels/nrs/whl/Publications/Journals/Gold-Simms_2000.pdf. Accessed 14 Mar 2011

Gold AJ, Stolt M, Rosenblatt AE, Groffman PM, Addy K, Kellogg DQ (2001) Landscape attributes as controls on ground water nitrate removal capacity of riparian zones. J Am Water Resour Assoc 37:1457–1464

Graf WL (1999) Dam nation: a geographic census of American dams and their large-scale hydrologic impacts. Water Resour Res 35:1305–1311

Granger S, Nixon S, Buckley B (2007) Procedures used in the construction of a denitrifying border. Rhode Island Coastal Resources Management Council Report, NA04NOS4190056

Groffman PM, Boulware NJ, Zipperer WC, Pouyat RV, Band LE, Colosimo MF (2002) Soil nitrogen cycle processes in urban riparian zones. Environ Sci Technol 36:4547–4552

Groffman PM, Dorsey AM, Mayer PM (2005) Nitrogen processing within geomorphic structures in urban streams. J N Am Benthol Soc 24:316–625

Groffman P, Butterbach-Bahl K, Fulweiler W, Gold A, Morse J, Stander E, Tague C et al (2009) Incorporating spatially and temporally explicit phenomena (hotspots and hot moments) in denitrification models. Biogeochemistry 93:49–77

Gruca-Rokosz R, Tomaszek JA, Koszelnik P (2009) Denitrification in the sediment of a eutrophic reservoir measured with the isotope pairing technique. Oceanol Hydrobiol Stud 38:75–81

Hammer DA (1989) Constructed wetlands for wastewater treatment: municipal, industrial and agricultural. Lewis, Chelsea

Hammer DA (1992) Designing constructed wetlands systems to treat agricultural nonpoint source pollution. Ecol Eng 1:49–82

Hammer DA, Knight RL (1994) Designing constructed wetlands for nitrogen removal. Water Sci Technol 29:15–27

Harrison J, Maranger R, Alexander R, Giblin A, Jacinthe P-A, Mayorga E et al (2009) The regional and global significance of nitrogen removal in lakes and reservoirs. Biogeochemistry 93:143–157

Harrison MD, Groffman PM, Mayer PM, Kaushal SS, Newcomer TA (2011) Denitrification in alluvial wetlands in an urban landscape. J Environ Qual 40:634–646

Hartranft JL, Merritts DJ, Walter RC, Rahnis M (2011) Big Spring Run restoration experiment: policy, geomorphology, and aquatic ecosystems in the Big Spring Run watershed, Lancaster County, PA. Sustain 24:24–30

He Y, Wilson JT, Wilkin RT (2008) Transformation of reactive iron minerals in a permeable reactive barrier (biowall) used to treat TCE in groundwater. Environ Sci Technol 42:6690–6696

Hernandez ME, Mitsch WJ (2007) Denitrification in created riverine wetlands: influence of hydrology and season. Ecol Eng 30:78–88

Hill B (1979) Uptake and release of nutrients by aquatic macrophytes. Aquat Bot 7:87–93

Hill AR (1996) Nitrate removal in stream riparian zones. J Environ Qual 25:743–755

Howarth R, Anderson D, Cloern J, Elfring C, Hopkinson C, Lapointe B et al (2000) Nutrient pollution of coastal rivers, bays, and seas. Issues Ecol 7:1–15

Hsieh CH, Davis AP, Needelman BA (2007) Nitrogen removal from urban stormwater runoff through layered bioretention columns. Water Environ Res 79:2404–2411

Hunt WF III, Jarrett AR, Smith JT, Sharkey LJ (2006) Evaluating bioretention hydrology and nutrient removal at three field sites in North Carolina. J Irrig Drain E 132:600–608

Hunter WJ (2001) Use of vegetable oil in pilot-scale denitrifying barrier. J Contam Hydrol 53:119–131

Jansson M, Andersson R, Berggren H, Leonardson L (1994) Wetlands and lakes as nitrogen traps. Ambio 23:320–325

Jordan SJ, Stoffer J, Nestlerode JA (2011) Wetlands as sinks for reactive nitrogen at continental and global scales: a meta-analysis. Ecosystems 14:144–155

Kadlec RH (1994) Wetlands for water polishing: free water surface wetlands. In: Mitsch WJ (ed) Global wetlands: Old world and new. Elsevier, Amsterdam, Netherlands, pp 335–349

Kadlec RH (2005) Nitrogen farming for pollution control. J Environ Sci Health A 40:1307–1330

Kadlec RH (2009) Comparison of free water and horizontal subsurface treatment wetlands. Ecol Eng 35:159–174

Kadlec RH, Wallace SD (2008) Treatment wetlands, 2nd ed. CRC Press, Boca Raton

Kasahara T, Hill AR (2006a) Effects of riffle-step restoration on hyporheic zone chemistry in N-rich lowland streams. Can J Fish Aquat Sci 63:120–133

Kasahara T, Hill AR (2006b) Hyporheic exchange flows induced by constructed riffles and steps in lowland streams in southern Ontario, Canada. Hydrol Process 20:4287–4305

Kaushal SS, Groffman PM, Mayer PM, Striz EA, Doheny EJ, Gold AJ (2008a) Effects of stream restoration on denitrification at the riparian–stream interface of an urbanizing watershed of the mid-Atlantic U.S. Ecol Appl 18:789–804

Kaushal SS, Groffman PM, Band LE, Shields CA, Morgan RP, Palmer MA et al (2008b) Interaction between urbanization and climate variability amplifies watershed nitrate export in Maryland. Environ Sci Technol 42:5872–5878

Kaushal SS, Groffman PM, Band LE, Elliott E, Kendall CA (2011) Tracking nonpoint nitrogen pollution in human-impacted watersheds. Environ Sci Technol 45:8225–8232

Keeney D (1986) Sources of nitrate to groundwater. CRC Crit Rev Environ Control 16:257–304

Kellogg DQ, Gold AJ, Cox S, Addy K, August PV (2010) A geospatial approach for assessing denitrification sinks within lower-order catchments. Ecol Eng 36:1596–1606

Kim H, Seagren EA, Davis AP (2003) Engineered bioretention for removal of nitrate from stormwater runoff. Water Environ Res 75:355–367

Klocker CA, Kaushal SS, Groffman PM, Mayer PM, Morgan RP (2009) Nitrogen uptake and denitrification in a restored urban stream. Aquat Sci 71:411–424

Lamb BE, Gold AJ, Loomis GW, McKiel CG (1990) Nitrogen removal for on-site sewage disposal: a recirculating sand filter/rock tank design. Trans ASAE 33:525–531

Lave R (2009) The controversy over natural channel design: substantive explanations and potential avenues for resolution. J Am Water Resour Assoc 45:1519–1532

LID Center (2011a) Bioretention. http://www.lid-stormwater.net/bio_costs.htm. Accessed 30 May 2011

LID Center (2011b) Permeable paver. http://www.lid-stormwater.net/permpaver_costs.htm. Accessed 30 May 2011

Lowrance R, Vellidis G, Hubbard RK (1995) Denitrification in a restored riparian forest wetland. J Environ Qual 24:808–815

Lowrance R, Altier LS, Newbold JD, Schnabel RR, Groffman PM, Denver JM et al (1997) Water quality functions of riparian forest buffers in Chesapeake Bay watersheds. Environ Manag 21:687–712

Lucas WC, Greenway M (2008) Nutrient retention in vegetated and nonvegetated bioretention mesocosms. J Irrig Drain E 134:613–623

Ludwig RD, Smyth D, Blowes DW, Spink LE, Wilkin RT, Jewett DJ et al (2009) Treatment of arsenic, heavy metals, and acidity using a mixed ZVI-compost PRB. Environ Sci Technol 43:1976–1979

Martin TL, Kaushik NK, Trevors JT, Whiteley HR (1999) Review: denitrification in temperate climate riparian zones. Water Air Soil Pollut 111:171–186

Mayer PM, Reynolds SK, McCutchen MD, Canfield TJ (2007) Meta-analysis of nitrogen removal in riparian buffers. J Environ Qual 36:1172–1180

Mayer PM, Groffman PM, Striz E, Kaushal SS (2010) Nitrogen dynamics at the groundwater/surface water interface of a degraded urban stream. J Environ Qual 39:810–823

Meyer JL, Paul MJ, Taulbee WK (2005) Stream ecosystem function in urbanizing landscapes. J N Am Benthol Soc 24:602–612

Mitchell MJ, Piatek KB, Christopher SF, Mayer B, Kendall C, McHale PJ (2006) Solute sources in stream water during consecutive fall storms in a Northern hardwood forest watershed: a combined hydrological, chemical and isotopic approach. Biogeochemistry 78:217–246

Mitchell CPJ, Branfireun BA, Kolka RK (2008) Assessing sulfate and carbon controls on net methylmercury production in peatlands: an in situ mesocosm approach. Appl Geochem 23:503–518

Mitsch WJ (1992) Landscape design and the role of created, restored, and natural riparian wetlands in controlling nonpoint source pollution. Ecol Eng 1:27–47

Mitsch WJ, Gosselink JG (2000) Wetlands, 3rd ed. Wiley, New York

Mitsch WJ, Day JW, Zhang L, Lane RR (2005) Nitrate-nitrogen retention in wetlands in the Mississippi River Basin. Ecol Eng 24:267–278

Moorman TB, Parkin TB, Kaspar TC, Jaynes DB (2010) Denitrification activity, wood loss, and N_2O emissions over nine years from a wood chip bioreactor. Ecol Eng 36:1567–1574

Nahlik AM, Mitsch WJ (2006) Tropical treatment wetlands dominated by free-floating macrophytes for water quality improvement in Costa Rica. Ecol Eng 28:246–257

Naiman RJ, Decamps H, McClain ME (2005) Riparia: ecology, conservation, and management of streamside communities. Elsevier Academic Press, London

Natural Resources Conservation Service (2011) USDA NRCS soil. http://www.soils.usda.gov. Accessed 9 Oct 2011

Newbold JD, Herbert S, Sweeney BW, Kiry P, Alberts SJ (2010) Water quality functions of a 15-year-old riparian forest buffer system. J Am Water Resour Assoc 46:299–310

North Carolina Department of Environment and Natural Resources (2005) Updated draft manual of stormwater best management practices. NCDNR, Raleigh

Oakley S, Gold AJ, Oczkowksi AJ (2010) Nitrogen control through decentralized wastewater treatment: process performance and alternative management strategies. Ecol Eng 36:1520–1531

Opperman JJ, Galloway GE, Fargione J, Mount JF, Richter BD, Secchi S (2009) Sustainable floodplains through large-scale reconnection to rivers. Science 326:1487–1488

Orr CH, Roth BM, Forshay KJ, Gonzales JD, Papenfus MM, Wassell RDG (2004) Examination of physical and regulatory variables leading to small dam removal in Wisconsin. Environ Manag 33:99–109

Palmer MA, Filoso S (2009) Restoration of ecosystem services for environmental markets. Science 325:575–576

Palone RS, Todd AH (1997) A Chesapeake Bay riparian handbook: a guide for establishing and maintaining riparian forest buffers. United States Department of Agriculture Forest service, NA-TP-02-97, Radnor

Parola AC Jr, Hansen C (2011) Reestablishing groundwater and surface water connections in stream restoration. Sustain 24:2–7

Passeport E, Hunt WF, Line DE, Smith RA, Brown RA (2009) Field study of the ability of two grassed bioretention cells to reduce storm-water runoff pollution. J Irrig Drain E 135:505–510

Puckett LJ (2004) Hydrogeologic controls on the transport and fate of nitrate in ground water beneath riparian buffer zones: Results from 13 studies across the United States. Water Sci Technol 49:47–53

Rabalais NN, Turner RE, Justic D, Dortch Q, Wiseman WJ Jr, Gupta BKS (1996) Nutrient changes in the Mississippi River and system response on the adjacent continental shelf. Estuaries 19:386–407

Rabalais NN, Turner RE, Wiseman WJ Jr (2001) Hypoxia in the Gulf of Mexico. J Environ Qual 30:320–329

Read J, Wevill T, Fletcher T, Deletic A (2008) Variation among plant species in pollutant removal from stormwater in biofiltration systems. Water Res 42:893–902

Reddy KR, Patrick WH (1984) Nitrogen transformations and loss in flooded soils and sediments. CRC Crit Rev Environ Control 13:273–309

Roberts BJ, Mulholland PJ, Houser JN (2007) Effects of upland disturbance and instream restoration on hydrodynamics and ammonium uptake in headwater streams. J N Am Benthol Soc 24:613–625

Roberts DC, Clark CD, English BC, Park WM, Roberts RK (2009) Estimating annualized riparian buffer costs for the Harpeth River watershed. Rev Agric Econ 31:894–913

Robertson WD (2010) Nitrate removal rates in woodchip media of varying age. Ecol Eng 36:1581–1587

Robertson WD, Cherry JA (1995) In situ denitrification of septic-system nitrate using reactive porous media barriers: Field trials. Ground Water 33:99–111

Robertson WD, Blowes DW, Ptacek CJ, Cherry JA (2000) Long-term of in situ reactive barriers for nitrate remediation. Ground Water 38:689–695

Robertson WD, Yeung N, van Driel PW, Lombardo PS (2005) High permeability layers for remediation of ground water: go wide, not deep. Ground Water 43:574–581

Robertson WD, Vogan JL, Lombardo PS (2008) Nitrate removal rates in a 15-year old permeable reactive barrier treating septic system nitrate. Ground Water Monit Remediat 28:65–72

Robertson WD, Ptacek CJ, Brown SJ (2009) Rates of nitrate and perchlorate removal in a 5-year-old wood particle reactor treating agricultural drainage. Ground Water Monit Remediat 29:87–94

Rosenzweig BR, Smith JA, Baeck ML, Jaffé PR (2011) Monitoring nitrogen loading and retention in an urban stormwater detention pond. J Environ Qual 40:598–609

Rosgen DL (1994) A classification of natural rivers. Catena 22:169–199

Rosgen DL (1996) Applied river morphology. Wildland Hydrology Books, Pagosa Springs

Sabater S, Butturini A, Clement J-C, Burt T, Dowrick D, Hefting M et al (2003) Nitrogen removal by riparian buffers along a European climatic gradient: patterns and factors of variation. Ecosystems 6:20–30

Saunders D-L, Kalff J (2001) Denitrification rates in the sediments of Lake Memphremagog, Canada—USA. Water Res 35:1897–1904
Schipper LA, Vojvodic-Vukovic M (1998) Nitrate removal from ground water using a denitrification wall amended with sawdust: Field trials. J Environ Qual 27:664–668
Schipper LA, Vojvodic-Vukovic M (2000) Rates of nitrate removal from groundwater and denitrification in a constructed denitrification wall. Ecol Eng 14:269–278
Schipper LA, Vojvodic-Vukovic M (2001) Five years of nitrate removal, denitrification and carbon dynamics in a denitrification wall. Water Res 35:3473–3477
Schipper LA, Barkle GF, Vojvodic-Vukovic M (2005) Maximum rates of nitrate removal in a denitrification wall. J Environ Qual 34:1270–1276
Schipper LA, Robertson WD, Gold AJ, Jaynes DB, Cameron SC (2010a) Denitrifying bioreactors—an approach for reducing nitrate loads to receiving waters. Ecol Eng 36:1532–1543
Schipper LA, Cameron SC, Warneke S (2010b) Nitrate removal from three different effluents using large-scale denitrification beds. Ecol Eng 36:1552–1557
Schultz RC, Colletti JP, Isenhart TM, Simpkings WW, Mize CW, Thompson ML (1995) Design and placement of a multispecies riparian buffer strip. Agroforest Syst 29:201–225
Seitzinger S (1988) Denitrification in freshwater and coastal marine ecosystems: ecological and geochemical significance. Limnol Oceanogr 33:702–724
Seitzinger S, Harrison J, Bohlke J, Bouwman A, Lowrance R, Peterson B et al (2006) Denitrification across landscapes and waterscapes: a synthesis. Ecol Appl 16:2064–2090
Selvakumar A, O'Connor TP, Struck SD (2010) Role of stream restoration in improving benthic macroinvertebrates and in-stream water quality in an urban watershed: case study. J Environ Eng 136:127–140
Shields CA, Band LE, Law N, Groffman PM, Kaushal SS, Savvas K et al (2008) Streamflow distribution of non-point source nitrogen export from urban-rural catchments in the Chesapeake Bay watershed. Water Resour Res 44:1–13
Siegrist RL, Jenssen PD (1989) Nitrogen removal during wastewater infiltration as affected by design and environmental factors. In: Proceedings of the 6th Northwest on-site wastewater treatment short course. University of Washington, Seattle, pp 304–318
Sivirichi GM, Kaushal SS, Mayer PM, Welty C, Belt K, Newcomer TA et al (2011) Longitudinal variability in streamwater chemistry and carbon and nitrogen fluxes in restored and degraded urban stream networks. J Environ Monit 13:288–303
Stander EK, Ehrenfeld JG (2009) Rapid assessment of urban wetlands: do hydrogeomorphic classification and reference criteria work? Environ Manag 43:725–742
Su C, Puls RW (2007) Removal of added nitrate in the single, binary, and ternary systems of cotton burr compost, zero-valent iron, and sediment: implications for groundwater nitrate remediation using permeable reactive barriers. Chemosphere 67:1653–1662
Sudduth EB, Hassett BA, Cada P, Bernhardt E (2011) Testing the field of dreams hypothesis: functional responses to urbanization and restoration in stream ecosystems. Ecol Appl 21:1972–1988
Tanner CC, Nguyen ML, Sukias JPS (2005) Nutrient removal by a constructed wetland treating subsurface drainage from grazed dairy pasture. Agric Ecosyst Environ 105:145–162
Teiter S, Mander U (2005) Emission of N_2O, N_2, CH_4 and CO_2 from constructed wetlands for wastewater treatment and from riparian buffer zones. Ecol Eng 25:528–541
Thurston HW, Roy AH, Shuster WD, Morrison MA, Taylor MA, Cabezas H (2008) Using economic incentives to manage stormwater runoff in the Shepherd Creek Watershed, Part I. United States Environmental Protection Agency Report No. EPA-600/R-08-129. National Risk Management Research Laboratory, USEPA, Cincinnati
United States Army Corps of Engineers (2011) National Inventory of Dams online database. http://www.nid.usace.army.mil. Accessed 13 Oct 2011
United States Environmental Protection Agency (2006) Baltimore County stream restoration improves quality of life. USEPA/903/F-06/008. http://www.epa.gov/bioiweb1/pdf/EPA-903-F-06-008 BaltimoreCountyStreamRestorationImprovesQualityofLife.pdf. Accessed 13 Oct 2011
United States Environmental Protection Agency (2011) Reactive nitrogen in the United States: an analysis of inputs, flows, consequences, and management options. A report of the EPA science advisory board, EPA-SAB-11-013
United States Geological Survey (2006) National elevation dataset. http://www.ned.usgs.gov. Accessed 13 Oct 2011
United States Geological Survey (2011a) NLCD 2006 product description provisional version. http://www.mrlc.gov/nlcd_2006.php. Accessed 13 Oct 2011
United States Geological Survey (2011b) National hydrography dataset. http://www.nhd.usgs.gov. Accessed 13 Oct 2011
Verhoeven JTA, Arheimer B, Yin C, Hefting MM (2006) Regional and global concerns over wetlands and water quality. Trends Ecol Evol 21:96–103
Vidon P, Hill AR (2006) A landscape based approach to estimate riparian hydrological and nitrate removal functions. J Am Water Resour Assoc 42:1099–1112
Vidon P, Allan C, Burns D, Duval T, Gurwick N, Inamdar S et al (2010) Hot spots and hot moments in riparian zones: Potential for improved water quality management. J Am Water Resour Assoc 46:278–298
Vymazal J (2005) Horizontal sub-surface flow and hybrid constructed wetlands systems for wastewater treatment. Ecol Eng 25:478–490
Vymazal J (2009) The use of constructed wetlands with horizontal sub-surface flow for various types of wastewater. Ecol Eng 35:1–17
Vymazal J, Greenway M, Tonderski K, Brix H, Mander U (2006) Constructed wetlands for wastewater treatment. Wetlands and natural resource management. Ecol Stud 190:69–96
Wall LG, Tank JL, Royer TV, Bernot MJ (2005) Spatial and temporal variability in sediment denitrification within an agriculturally influenced reservoir. Biogeochemistry 76:85–111
Walsh CJ, Roy AH, Feminella JW, Cottingham PD, Groffman PM, Morgan IIRP (2005) The urban stream syndrome: current knowledge and the search for a cure. J N Am Benthol Soc 24:706–723
Walter RC, Merritts DJ (2008) Natural streams and the legacy of water-powered mills. Science 319:299–304
Water Environment Research Foundation (2010) Cost of individual and small community wastewater management systems. http://www.werf.org/AM/Template.cfm?Section=Decentralized_Systems&Template=/CM/ContentDisplay.cfm&ContentID=15581. Accessed 4 Mar 2011
Welsch DJ (1991) Riparian forest buffers, function and design for protection and enhancement of water resources. United States Department of Agriculture Forest Service Publication No. NA-PR-07-91. USDA, Radnor. http://www.na.fs.fed.us/spfo/pubs/n_resource/buffer/cover.htm. Accessed 13 Oct 2011
Wenger SJ, Roy AH, Jackson CR, Bernhardt ES, Carter TL, Filoso S, e al. (2009) Twenty-six key research questions in urban stream ecology: An assessment of the state of the science. J North Am Benthol Soc 28:1080–1098
Wilkin RT, Acree SD, Ross RR, Beak DG, Lee TR (2009) Performance of a zero-valent iron reactive barrier for the treatment of arsenic in groundwater: Part 1. Hydrogeochemical studies. J Contam Hydrol 106:1–14
Zhang X, Liu X, Zhang M, Dahlgren RA, Eitzel M (2010) A review of vegetated buffers and a meta-analysis of their mitigation efficacy in reducing nonpoint source pollution. J Environ Qual 39:76–84